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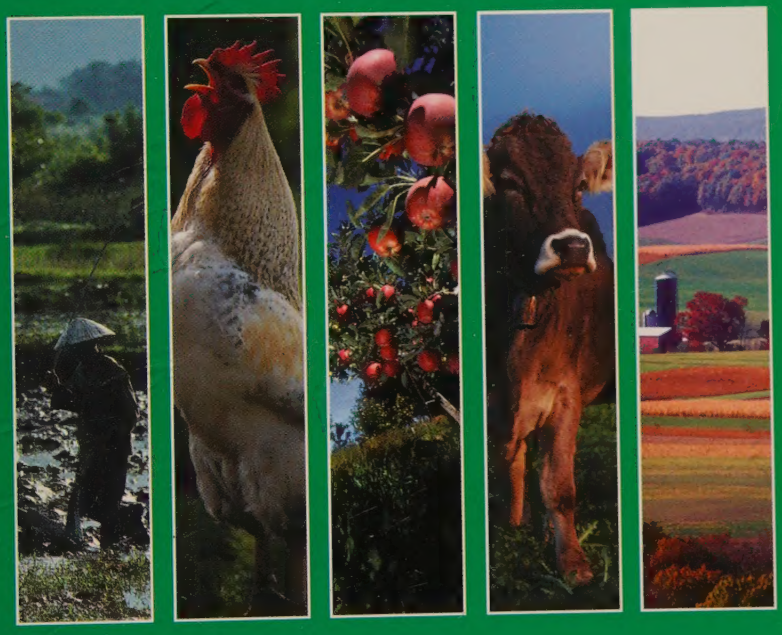

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AMERICAN JOURNAL OF ALTERNATIVE AGRICULTURE

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The American Journal of Alternative Agriculture provides an international forum for original research articles on the biological, physical, social science and policy aspects of alternative agriculture. The journal particularly focuses on multidisciplinary studies and covers the character and potential of alternative, sustainable and organic farming systems.

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AMERICAN JOURNAL OF ALTERNATIVE AGRICULTURE

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Editorial

The theme of the papers in this issue addresses the potential for using farmland and forest management for long-term removal of significant quantities of carbon dioxide (CO_2), a major greenhouse gas, from the atmosphere. The rising levels of CO_2, primarily from energy production worldwide, have been linked to global warming that, if allowed to continue, may eventually adversely impact food supplies and create major environmental hazards such as increased severity of drought and wind erosion, and rising ocean levels. Long-term sequestration of C in agriculture by crops and trees, through its retention and conversion in the soil to organic matter, helps to enhance soil quality that, in turn, increases plant productivity and decreases erosion by wind and water. It is well known that the organic matter content of soils can be increased through selective crop and plant residue management along with reduced tillage or, better yet, no-till farming practices. The same holds true in the management of agroforestry and forest systems. Sustainable agriculture, which is ecologically based and well known for its environmental enhancement, is strongly dependent on practices that promote C conservation in the soil; thus, C sequestration is a major benefit of these farming systems.

These five papers, with their lead-off introduction, explore various prospects for increasing C sequestration in agricultural, agroforestry and forest systems through changes in traditional management practices under diverse land-use settings. Considerable emphasis is given to the incentives required under different economic conditions to induce land managers to participate in an effective C sequestration program. These include a comparison of payments to US farmers to change management practices to conserve C on lower- versus higher-producing soils, and payments that may induce farmers to re-establish forest cover on often flooded bottomlands that have been cleared for marginally yielding soybean production. Two papers from Indonesia deal with the costs of conserving the C sequestered in the remaining pristine and community-managed forests, and emphasize the need for removing current policy disincentives for growing trees. These papers also provide valuable information on C measurement protocols, and, as stated by Carpentier in her introductory paper, 'coverage of . . . the real, additional, verifiable, and permanent issues to qualify as ''certifiable emission reduction''.' Finally, a paper on the Canadian perspective investigates the effectiveness of several management alternatives for increasing C sequestration on agricultural lands and, if applied, their overall impact on the national economy.

Finally, we wish to express our thanks and appreciation to Dr Chantal Line Carpentier for her outstanding leadership in organizing this special issue, for her own lead introduction paper, and to the authors of the five papers that have presented some of the most advanced conceptual and real-world perspectives on the effectiveness and feasibility for sequestering C on agricultural and forestry lands.

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Carbon sequestration in agriculture and forestry to offset carbon emissions and achieve sustainable production systems

Chantal Line Carpentier

The goal of this issue of the *AJAA* on carbon (C) sequestration in agriculture and forestry is to demystify the potential for the agroforestry sectors in Annex 1 countries of the Kyoto Protocol (exemplified by the US and Canada in this issue) and non-Annex 1 countries (exemplified by Indonesia) to be a source of inexpensive C sinks to offset C emissions from other sectors, while at the same time benefiting small farmers and conservation.

Sustainable agriculture (incorporating little to no chemical inputs, polyculture, integrated pest management, etc.) and managed forestry offer many environmental benefits over conventional agriculture and forestry practices. In an ideal world, the market prices of the outputs produced in a sustainable manner would reflect these benefits. As a result, when buying tomatoes (*Lycopersicon esculentum*), for example, we would pay not only for the costs of production (including a reasonable return to the farmer), but also for the environmental benefits, such as biodiversity, forest conservation, and soil and water quality protection, that resulted in the production of the tomatoes. One benefit that can result from sustainable agriculture is C sequestration.

With the development of the Kyoto Protocol, many thought that the possibility of earning valuable C credits through C sequestration could become an important incentive to encourage activities that not only absorbed carbon, but which, in an attempt to absorb carbon, also encouraged the conservation of remaining tropical forests. These hopes, however, have been put to an end, at least for while, due to the recent exclusion of agriculture, forest management and conservation as eligible C reduction activities in the Clean Development Mechanism (CDM). Even though the Indonesian papers were written before that decision was taken, they still provide a good overview of the potential for C sequestration from land management, its associated costs, the additional benefits that might be generated, and the challenges of administering and organizing a system that would be able to quantify the sequestration of C by millions of farms. Also, forest management and C offsets in agricultural soils are allowed for offset by Annex 1 countries [and in the Joint Implementation (JI)]. Experience and advances in measurement and monitoring in Annex 1 countries may also make it possible to allow these offsets in CDM initiatives of future commitment periods.

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In addition to presenting examples of C sequestration potential and prices at which farmers might be willing to change their current practices in different countries, and even within regions in the US, the five papers in this issue attempt to shed light on:

- C sink measurement protocols and problems in biomass, soil and land-use management across the world;
- C modeling examples to support measurement;
- a comparison of costs of sequestering C in agriculture to those in other industries; and
- the potential role of C sequestration in sustainable agriculture (and support to small-scale agriculture in the US and around the globe).

Before summarizing the results of each paper, it is important to review what C sequestration is (and is not). The Kyoto Protocol was launched at the Third Conference of the Parties (COP 3) to the UN Framework Convention on Climate Change held in Kyoto in 1997. Among the main outcomes of that agreement was the commitment by all Annex I countries to lower their emissions of six greenhouse gases—carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), HFCs, PFCs and SF_6—by at least 5% below 1990 levels, with a target date of between 2008 and 2012. Member states of the European Union, Switzerland and the Czech Republic, Bulgaria, Estonia, Romania and Slovakia committed to reduce their net greenhouse gas (GHG) emissions by 8%. Net emissions are calculated as emissions minus sinks and avoided emissions above and beyond what would have happened without the Protocol (referred to as the baseline). The US committed to lower its net emissions by 7% and Canada, Hungary, Poland and Japan by 6%. Some other countries are allowed to increase their emissions above the 1990 levels: these include Australia (8%), Iceland (10%) and Norway (1%) while others must stabilize them, including Russia, New Zealand and Ukraine. The six gases are translated into CO_2 equivalent (one Mg of C is equivalent to 3.67 Mg of CO_2) and will need to be reduced by the commitment reduction for the 2008–2012 period (average over the 5 years). The literature often refers to cap or target reduction, which is a country's 1990 CO_2-equivalent emissions minus the committed reduction. The cap for the US would be 93% of its 1990 emissions.

Land use in industrialized countries appears in two important articles of the Protocol. Article 3.4 stipulates that activities in the land use, land-use change and forestry (LULUCF) sector, that remove C from the atmosphere (afforestation, reforestation, the prevention of deforestation)—referred to as C sink or sequestra

tion—can be used to offset emission caps. Changes in these activities that release C into the atmosphere, like deforestation, are added to the total emission under Article 3.3.

Growing vegetation is a C sink because it removes CO_2 from the air and stores it, through photosynthesis, in plant biomass, both above and below ground. New C is sequestered until the vegetation matures, at which point the stock of C reaches equilibrium and ceases to grow anymore—forests are thus a world reservoir of C. As a forest is cleared for agriculture and pasture, this C is released into the atmosphere.

Worldwide, forests are a net source of C as C sequestered through afforestation and reforestation does not keep up with emission from the burning and clearing of forests, and timber extraction. Agriculture accounts for 20% of the world's human-enhanced GHG effects, but 50% of methane production and 70% of NO_x production (mainly through intensive livestock farming, wet rice-cultivation and fertilizer use). Farm and forest management to reduce these emissions, and increase C storage in the soil and wood, do exist, but for many reasons (including economic ones) are not adopted. For instance, agricultural soils are now net sources of CO_2 but could be net sinks for C through the use of composting or animal manure, low or no-till farming practices, and better timing of fertilizer application and plant uptake.

Carbon sequestration in developing (non-Annex 1) countries could also contribute to a reduction of GHGs in the atmosphere through the CDM that allows Annex 1 (and non-Annex 1) countries to obtain C credits by investing in sustainable development, carbon-reducing projects in non-Annex 1 countries (Article 12 of the Protocol). The CDM could benefit developing countries because of the financial and technical support they would receive, and they could also benefit from the resulting global reduction in GHG.

CDM could also benefit developed countries because the marginal costs of GHG abatement in these countries—the source of the majority of GHG emissions—is much higher than in economies in transition or developing countries. By way of illustration, some estimates show that the cost per Mg of C emission reductions in industrialized countries is in the vicinity of US\$5–10 Mg^{-1} compared to US\$5 Mg^{-1}, or less, in developing countries, and that many offset opportunities are available. This is due to the fact that C credits are currently estimated to be worth US\$3 Mg^{-1} C, and estimated to increase to US\$21 Mg^{-1} by 2010. Annex 1 country prices could reach US\$78 Mg^{-1} by the same period (given the withdrawal of the US and the economic collapse in Russia and the Ukraine, these numbers are now very uncertain and most likely overestimated).

This type of international collaboration could allow countries to achieve environmental objectives at lower costs than would otherwise be possible if each country were to achieve their GHG emissions independently, and could provide the needed capital to stimulate sustainable development initiatives in developing countries. [Two other market mechanisms can be used by Annex I countries. These are the Joint Implementation (JI) outlined in Article 6 and International Emissions Trading (IET) outlined in Article 17; see www.UNFCCC.de for details.] Some countries, known as the Umbrella group, have advocated the inclusion of C stored in forests as a source of C offsets because, in addition to storing the most C, forests have value to national and international stakeholders because of their role in modifying local climate, for watershed protection and for the maintenance of biodiversity.

For small farmers to benefit from the emerging C market, four conditions must be fulfilled. First, farmers must be aware of the development of the market, ongoing C prices and how they can satisfy the requirements for the market. C sequestration must, of course, first be approved as an offset activity. Second, emission reductions must be measurable, verifiable and rules must be applied consistently. Third, farmers must be willing to participate. That is, income from alternative land uses and management practices that sequester C, plus C payments, must be at least as large as income generated from current practices. Fourth, the transaction costs to deal with thousands of small farms must be reduced through innovative institutional arrangements. For society to benefit from emerging markets for C sequestration, they must be the most cost-effective policies for removing GHGs from the atmosphere and they must be politically acceptable.

Regarding the first condition, few options remain for non-Annex 1 countries. When papers in this issue were being written, it was uncertain whether C sequestration from forest management and in agricultural soils would qualify under the CDM. Since then, at the seventh Conference of the Parties (COP7) it was decided that only afforestation and reforestation would be allowed (although definitions of these terms have yet to been agreed upon) in the first round of commitment. C sequestration in agricultural soils, from forest management or forest conservation in developing countries would not be eligible. These C offsets were not included partly due to the second condition: measurement, additionality (baseline and leakage), non-permanence, and socio-economic and environmental issues (see Tomich et al. for a discussion of these issues). Under Article 3.4, Annex 1 countries can offset C emissions through agricultural soils and forest management (although the amount of offsets from forest management is limited).

With regard to the third condition, three papers, two for the US and one for Indonesia, present methodological and empirical evidence to support whether, and under what conditions, farmers would be willing to participate in a C sequestration scheme. Antle et al. estimate the costs of sequestering C on US Montana and Iowa farmland, emphasizing the impact of both biophysical and economic factors on the ability and willingness of farmers to change practices in order to sequester C. They demonstrate that it is more beneficial for farmers in lower economic opportunity areas of Montana to accept C payments to change their management than farmers in the higher producing soils of Iowa. A minimum of US\$10 Mg^{-1} of C would be necessary to induce changes in land use on low-opportunity lands, and even at US\$25 Mg^{-1}, the two states would only offset 0.5M Mg yr^{-1}.

Shabman et al. chose a low economic opportunity area of the US, the southern portion of the Mississippi Delta, and estimate the C payments that would be necessary to induce farmers to switch their often-flooded soybean (*Glycine max* L.) areas to bottomland forest. A minimum payment of US\$20 Mg^{-1}

(including soil C) would be necessary if timber value is not included as a source of income, and US$14 Mg^{-1} if timber value is included. However, COP7 has also limited the extent to which C emission can come from agricultural soil and forest management.

Tomich et al. estimate the C payment that would induce Indonesian farmers to conserve the C sequestered in their remaining pristine and community-managed forests. C payments necessary to maintain community-managed forests range from US$0.38 to US$20.57 Mg^{-1} if timber value is not included, and US$15–17 Mg^{-1} if timber value is included. It would cost less to maintain stored C in remaining primary forests; but neither of these would be eligible under CDM—as mentioned above, only afforestation and reforestation would be eligible after COP7.

The C measurement issue is covered in more detail in Roshetko et al. This paper covers the measurement issue in a case study of C sequestration in Indonesian homegardens. These latter two papers evaluate the potential applications of CDM in Indonesia. Thomich et al. also provide good coverage of measurement issues and the real, additional, verifiable and permanent issues to qualify as 'certifiable emission reduction'. In addition to providing recommendations on how to reduce the costs to small land owners around the world to enable them t have access to C reduction investment, they recommen removing two major barriers that reduce the disincentives t grow trees; i.e., tenure insecurity and trade, and marketin restrictions that prevent selling any trees. They contend that th Protocol may not lead to the most cost-effective ways o achieving sustainable forestry, since removing policy distortion may be the most cost-effective way to sequester C, but that i may not pass the additionality criteria of the protocol. Clearly stable and clear signals, including policies to reduce pric distortions and subsidies to C-loaded technologies, could go long way toward the adoption of cost-efficient energy, transport agricultural and other markets, and these may not be eligible fo credit under the Protocol.

A final paper by Thomassin presents the methodologica approach and results of a Canadian attempt to find the most cost effective way to achieve its agricultural sector's part of th commitment. The only practice that generates positive macro economic returns is the option to improve forage quality by 15% but it is found that this would only reduce emissions by 0.07% inclusive of soil sinks. All other alternatives, it was found, woul negatively impact the economy.

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A comparative examination of the efficiency of sequestering carbon in US agricultural soils

John M. Antle, Susan M. Capalbo, Siân Mooney, Edward T. Elliott, and Keith H. Paustian

Abstract. *This paper demonstrates the need to integrate biophysical and economic data to assess the competitiveness of US agriculture to provide soil carbon (C) and participate in a market for C credits. The paper discusses alternative methods of calculating the costs of soil C sequestration and compares the cost of sequestering soil C in Iowa and Montana. Results indicate that the opportunity cost per Mg of sequestered C varies in response to regional resource endowments and net returns. Economic models show that Montana could sequester a relatively small amount of soil C annually at a lower opportunity cost per Mg than Iowa, but Iowa can sequester larger quantities more efficiently. These results are compared with estimates of the cost of C sequestration from other domestic and international studies, and suggest that US agriculture could be competitive in domestic or international markets for C reduction credits.*

Key words: carbon credits, carbon reduction credits, carbon sequestration, economics of carbon sequestration, soil carbon storage, US agriculture

Introduction and Background

The Kyoto Protocol of the United Nations Framework Convention on Climate Change (UNFCCC, 1998) aims to reduce atmospheric concentrations of anthropogenic greenhouse gases (GHG) to minimize the potential for global climate change. Under the protocol, the US is required to reduce net emissions of GHG to 93% of 1990 levels between the years 2008 and 2012 (UNFCCC, 1998). This target could be met by reducing emissions of important GHG, principally N_2O (nitrous oxide), CH_4 (methane) and CO_2 (carbon dioxide), as well as removing CO_2 from the atmosphere and sequestering carbon (C) in agricultural or forest sinks. To facilitate these reductions the Kyoto protocol allows for the development of a trading scheme in emissions reductions credits. If C can be sequestered in agricultural soils at a lower cost than emissions reductions or sequestration efforts by other sectors, agricultural producers will be in a position to benefit from this new market opportunity.

Total US emissions of GHG are estimated at between 1485 and 1709 million Mg of C equivalent per year (MMgCE yr^{-1}), of which emissions from agriculture are thought to be between 109 and 123 MMgCE yr^{-1} (Lal et al., 1998). With appropriate management practices, agricultural soils can sequester additional C. Estimates suggest that up to 8% of US emissions can be reduced by sequestering C in cropland soils (Lal et al., 1998). Alternatively, US cropland has the potential to sequester between 4000 and 6000 MMg C if soil C can be returned to pre-cultivation levels (IPCC, 1996; Lal et al., 1998).

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In addition to domestic reductions in GHG, the Kyoto Protocol (Article 12) allows developed countries (those listed in Annex 1 of the Convention) to purchase emissions reduction credits from projects in developing countries and use these credits to offset their obligations to reduce GHG (UNFCCC, 1998). Developed countries can also trade emissions reduction credits between themselves. If US agriculture wishes to be a participant in a global market for C credits, it must sequester C at a cost that is competitive both nationally and internationally with other sources of C.

This paper demonstrates how biophysical and economic data can be integrated to examine the costs of supplying soil C. Studies by Antle et al. (2001b) and Pautsch et al. (2001) are used to compare the relative economic efficiency of states within the Northern Great Plains and Cornbelt regions at supplying soil C within a market for C credits. This is the first detailed comparison of the relative costs of soil C sequestration across different agro-ecozones of the US. We show that given the range of practices considered within each study, these agro-ecozones are not equally efficient at sequestering soil C since the opportunity cost per Mg of C sequestered varies spatially in response to economic and resource variability. Finally, we compare these estimates to the costs of C sequestration in other domestic and international projects, to examine whether agricultural producers in these agro-ecozones could be competitive in a domestic or international market for C reduction credits.

Biophysical Potential to Sequester C within US Cropland Soils

As land is converted from native vegetation to modern agriculture, C stored within the soil is oxidized and released into the atmosphere; in some cases, biomass production also decreases, reducing inputs of C into the soil (IPCC, 1996; Lal

et al., 1998). Tiessen et al. (1982), Mann (1986) and Rasmussen and Parton (1994) estimate that 20–50% of soil C is lost during the initial 20–50 years of cultivation. Because of this past depletion of soil C levels, cultivated soils in many areas have the capacity to store more C than they do at present (Lal et al., 1998).

Soil C can be increased by adopting practices that reduce soil disturbance and thus C oxidation and/or by increasing the amount of biomass produced on an area. As plants photosynthesize, they take CO_2 from the atmosphere and deposit it in the soil in the form of C stored in roots and other vegetative material. The inherent potential of a site to sequester soil C is influenced by biophysical factors such as soil type, climate, topography and vegetation. Factors that influence soil C emissions and sequestration are discussed in Lal et al. (1998, p. 22).

Historic and current land use and management can influence the C sequestration potential of cultivated sites. There are several land-use and management practices that can be adopted to increase soil C (CAST, 1992; Lal et al., 1998). For example, a switch from conventional to conservation tillage reduces C oxidation and thus emissions of CO_2. Switching from a crop-fallow system to continuous cropping increases biomass production and inputs of C into the soil (Lal et al., 1998). The potential for these practices to sequester additional C varies spatially, as shown in this paper, and is further influenced by the accompanying cropping intensity or tillage practice. Soil C is expected to increase relatively slowly during the first 2–5 years of management modifications, with larger changes between 5 and 10 years and flattening off thereafter until reaching a finite limit over a period of 50–100 years (Franzluebbers and Arshad, 1996; IPCC, 1996). If producers subsequently revert to their original practices, the stored C may be returned to the atmosphere.

To date there are few estimates of the potential for different agro-ecozones within the US to sequester additional soil C. The presence of spatially heterogeneous biophysical and economic conditions suggests that a single land-use or management practice will not be equally efficient at sequestering C at different sites.

The majority of cropland in Iowa (one of the Cornbelt states) is continuously cropped (Mitchell et al., 1997) and consequently has little potential to sequester additional soil C through an increase in biomass production. In these areas, research has focused upon reducing tillage intensity to increase the rate of soil C accumulation. In contrast, Montana (in the Northern Great Plains) is mostly in a crop-fallow rotation and research has concentrated upon intensifying crop production to sequester additional soil C. An increase in biomass production coupled with a decrease in tillage intensity could further increase the rate of C accumulation. An exhaustive list of potential management practices has not yet been examined in every area.

The potential annual increases in soil C per ha attributable to changes in tillage practices within Iowa and increased cropping intensity within Montana are presented in Table 1. These estimates are generated by Century, a C dynamics model

Table 1. Annual increase in soil C from management changes Iowa and Montana.

Change in management	Additional carbon[1] (Mg ha^{-1} yr^{-1})
Iowa—change in tillage practice	
Intensive tillage to moderate tillage	0.19
Intensive tillage to no-till	0.52
Montana—change in cropping system	
Spring wheat[2] fallow to continuous spring wheat	0.38
Spring wheat fallow to continuous winter wheat	0.44
Permanent grass to continuous spring wheat	0.06
Permanent grass to continuous winter wheat	0.13

[1]Estimates were calculated using Century (Parton et al., 1987; Paustian et al., 1992, 1996) and provided by the Natural Resource Ecology Laboratory, Colorado State University, Ft. Collins.
[2]*Triticum aestivum* L.

(Parton et al., 1987; Paustian et al., 1992, 1996). Table 1 shows that a change from conventional tillage to no-till can sequester an additional 0.52 MgC ha^{-1} yr^{-1} in Iowa, exceeding the potential to sequester additional C by increasing cropping intensity in Montana by 0.08 MgC ha^{-1} yr^{-1} (approximately 19%). The 1997 census of agriculture reports that there are 11 million ha of cropland in Iowa and 7 million ha of cropland in Montana (USDA, 1997). The combination of census figures and C rates from the Century model suggests that if Iowa producers switched to conservation tillage, they would sequester C at a faster rate per ha as well as sequestering a greater total C amount when compared to the potential to sequester C by increasing biomass production in Montana. However, simply considering the biophysical potential of an area to sequester soil C ignores economic, social and technical constraints. These factors will influence the costs of placing land in activities that sequester soil C and the willingness of producers to change their production practices. The role of economic constraints is discussed in the next section.

Economic Considerations

Incentive payments to producers

In this section we show that if farmers are making land-use and management decisions to maximize their current net returns they will obtain lower net returns when they change practices to sequester additional C. This reduction in returns is the opportunity cost to the farmer (producer) of sequestering soil C. The opportunity cost per Mg C will vary spatially with land productivity and production choices (Antle et al., 2001a).

If we assume that the goal of individual producers is to maximize their profits from production, they will be engaged currently in land-use and management practices that yield the highest net returns. Producers will change these management practices to those that sequester additional C only if they are provided with an incentive that compensates them for any economic losses.

There are many ways to structure economic incentives that encourage producers to participate in activities that sequester soil C (Antle and Mooney, 2001). The following example is based on a market scheme that pays producers for each Mg of additional C (see Antle et al., 2001b for more details). This scheme is analogous to a per ha payment that varies spatially in response to the ability of each site to sequester soil C (discussed in the next section). We assume that: production decisions on each ha are independent; participation is voluntary in any program designed to sequester additional C; measuring, monitoring and transactions costs are zero; and there are no social or technical constraints that limit the adoption of management practices that sequester soil C. Let π_0 represent profits per ha from current activities, and π_a profits from a management change that sequesters additional C, such that $\pi_0 > \pi_a$ and $\pi_0 - \pi_a$ is the opportunity cost per ha of sequestering additional C. Further let p represent the market price per Mg C paid to producers and let ΔC represent the additional C sequestered on that ha annually as a result of adopting the new management practice. Under the assumptions of this model, a producer will adopt a practice that sequesters additional soil C if $\pi_0 - \pi_a \leq p\Delta C$, that is if the total payment received per ha ($p\Delta C$) compensates the producer for the opportunity cost per ha incurred by changing management practices. Rearranging the previous expression, we obtain $\frac{\pi_0 - \pi_a}{\Delta C} \leq p$, which indicates that the price per Mg must be large enough to compensate the producer for the opportunity cost of sequestering each Mg of C before a producer will change management practices. The opportunity cost per Mg, $\frac{\pi_0 - \pi_a}{\Delta C}$, is determined by the quantity of C sequestered per ha as a result of changing management practices and the opportunity cost per ha of changing management practices. This expression supports the earlier assertion that the absolute quantity of C that can be sequestered on each ha annually, ΔC, does not reflect producer willingness to engage in activities that sequester additional soil C. Thus the biophysical potential of an area to sequester C and the economic costs of changing land management practices must be considered jointly.

Efficiency of Soil C Sequestration within Different Agricultural Regions of the US

Comparison of Montana and Iowa costs based on a per Mg scheme

The previous sections demonstrate that biophysical factors determine the ability of an area to sequester C, and economic factors determine the opportunity costs of changing management practices to sequester additional C. Antle et al. (2001b) and Pautsch et al. (2001) are two studies that have integrated biophysical and economic data to examine the efficiency of soil C sequestration for different regions of the US. Both studies can be compared to show how the cost per Mg C varies over space.

Pautsch et al. (2001) use a discrete-choice econometric logit model to examine producer decision-making in response to payments offered to sequester soil C. Producers are offered variable per acre (1 acre = 0.405 ha) payments (analogous to constant per Mg payments) for additional Mg C produced by a change from conventional to conservation tillage.

Fourteen major rotations were examined, based on production of corn (*Zea mays* L.), soybean (*Glycine max* L.), wheat (*Triticum aestivum* L.), sorghum (*Sorghum* sp.) and hay. Corn and soybean were the prevalent crops, planted on 90% of the land area. Producers make a decision to switch tillage practices if the expected net returns under the new tillage system, plus payments for the additional C sequestered, are maximized. The probability of adopting a conservation tillage practice varies across production sites in response to biophysical and economic conditions. Annual site-specific changes in C are estimated using a meta-model developed by Mitchell et al. (1997) based on EPIC (Erosion Productivity Impact Calculator). Carbon changes are estimated over 30 years (Mitchell et al., 1997) and converted to an annual basis assuming a linear accumulation over time for approximately 9 million ha of cropland soils. Site-specific annual changes in soil C are aggregated across sites to obtain the aggregate C supply function for Iowa corresponding to a per Mg payment scheme for C produced by a change in tillage practices.

Antle et al. (2001b) examine the economic potential for C sequestration on approximately 3 million ha of cropland soils within six agro-ecozones in Montana. A site-specific econometric-simulation process model (Antle and Capalbo, 2001) is used to predict changes in cropping systems in response to payments offered for each Mg of C produced as a result of intensifying biomass production. Analogous to Pautsch et al. (2001), producers are offered payments for each additional Mg of C that they can sequester by changing their cropping systems from either a crop–fallow system to grass or continuous cropping. Seven systems are considered by the model, spring wheat–fallow, barley (*Hordeum vulgare* L.)–fallow, winter wheat–fallow, grass and spring wheat, barley and winter wheat continuous cropping. The additional soil C expected from a change in crop systems is estimated by the Century model (Parton et al., 1987; Paustian et al., 1992, 1996). Carbon is assumed to accumulate over 20 years at a linear rate in accordance with the soil science literature (Mitchell et al., 1997; Watson et al., 2000). If the returns from the new system plus payments for the additional C produced exceed the returns from the original system, the producer will change cropping systems (consistent with the theoretical framework presented earlier within this paper).

In these two studies the costs of sequestering soil C are considered from different viewpoints. Antle et al. (2001b) report the marginal costs of supplying soil C within six agro-ecozones over a period of 20 years. Marginal cost (MC) is the increase in total producer opportunity cost as a result of sequestering one additional unit of C. In contrast, Pautsch et al. (2001) provide an estimate of the average annual cost of purchasing soil C. This is the cost incurred by a government or private agent when they purchase C, i.e., the price per Mg multiplied by the Mg C purchased, and will exceed the opportunity costs incurred by producers generating C in most cases. The cost estimates per Mg C from Antle et al. (2001b) and Pautsch et al. (2001) can be compared directly by converting them to a common metric. We

will show that the marginal cost of sequestering a given quantity of C is approximately equal to the average cost of purchasing that same quantity and thus can be compared directly.

The total producer opportunity cost of supplying a given quantity of C is the area under the marginal cost curve; for example, the opportunity cost of supplying C_1 Mg of C is area A (Fig. 1). The marginal cost of sequestering one more Mg for a total of C_2 Mg of C is area B. P_2 is the price paid for each Mg of C at quantity C_2, area B can be approximated as $MC_2 = P_2(C_2 - C_1) = P_2$ (where $C_2 - C_1 = 1$). The total cost of purchasing C_2 Mg of C by the government or other purchasing agent is $P_2{*}C_2$ and the average cost per unit is equal to $(P_2{*}C_2)/C_2 = P_2$ or the MC of the last Mg of C. Figure 1 and the previous discussion demonstrate that the marginal cost of supplying the last unit of a given quantity of C is approximately equal to the average cost per unit of purchasing that same quantity. Thus, the estimated costs of C sequestration generated by Pautsch et al. (2001) can be compared directly to Antle et al. (2001b) after the latter estimates have been converted to an annual basis.

The linear increase in C over 20 years used by Antle et al. (2001b) was converted to an annual basis by dividing through by 20. The annual quantity of C supplied at each price level within each of the six agro-ecozones was then summed to create an annual supply curve for Montana. Both the costs and C quantities are comparable to the data of Pautsch et al. (2001) in that they represent the cost per Mg C from changes in consumer decision-making and management practices in response to an annual per Mg C payment.

One remaining difference between the two studies is the model used to estimate changes in soil C. Antle et al. (2001b) use Century while Pautsch et al. (2001) use a meta-model (Mitchell et al., 1997) derived from the EPIC model. Mitchell et al. (1997) show that an additional 0.2 Mg ha^{-1} yr^{-1} C can be sequestered in Iowa soils by changing from conventional to conservation tillage. This figure was calculated from Tables 1, 3 and 4 and associated discussions in Mitchell et al. (1997). It is similar to the estimate from Century (Table 1) for a switch from conventional to moderate tillage but much less than that predicted by Century for a switch from conventional tillage to no-till. These figures suggest that if Pautsch et al. (2001) had used estimates from Century, the potential for soil C accumulation on each hectare in Iowa might be greater than reported, and demonstrates the potential for different outcomes under alternative modeling combinations.

Figure 2 compares the marginal cost (MC) per Mg C in Montana (MC_M) and Iowa (MC_I) under payments that range between \$10 and \$700 Mg^{-1} C. The relative placement of these curves is a function of both the spatial differences in the MC of soil C sequestration and the size of each study area. Both curves are inelastic throughout the price range considered, meaning that an increase in the price offered for each Mg of C will result in a less than proportional increase in the quantity of C supplied in both areas. At prices greater than \$340 Mg^{-1}, MC_M is almost vertical (or perfectly inelastic), implying that further price increases will generate negligible increases in C. This occurs because most of the cropland has already switched to management practices that sequester soil C. Iowa producers can sequester a greater total amount of C, in part reflecting the larger area of cropland suitable for soil C sequestration.

Figure 2 also shows that the two regions are not equally efficient at sequestering soil C. At prices less than \$155 Mg^{-1}, C can be sequestered more efficiently in Montana cropland soils, because a given quantity of C can be supplied at a lower marginal cost. For example, 0.5 million MgC yr^{-1} can be supplied at \$30 Mg^{-1} in Montana and \$80 Mg^{-1} in Iowa. The total cost to producers is represented by the area under the marginal cost curves to the left of this quantity. MC_I lies above MC_M at prices less than \$155 Mg^{-1}, encompassing a larger area at any given C level. Montana producers would supply more C at prices less than \$155 Mg^{-1} because the total producer costs of supplying C over this price range are higher for producers in Iowa than those in Montana. However, at higher prices, Montana's available land becomes a limiting factor, so the opportunity cost per Mg rises rapidly and Iowa producers become more efficient at sequestering soil C. Sandor and Skees (1999) estimate that the market price of C credits could range

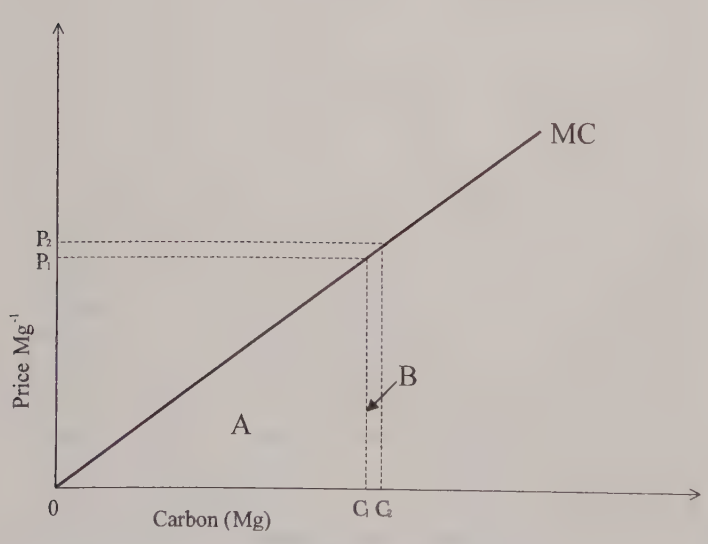

Figure 1. Marginal cost (MC) of carbon production.

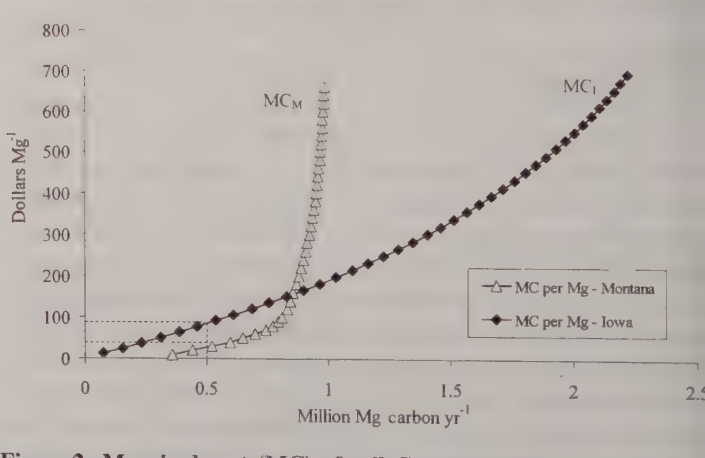

Figure 2. Marginal cost (MC) of soil C sequestration under a per payment scheme implemented in Iowa and Montana.

between \$15 and \$348 Mg^{-1}. Figure 2 demonstrates that C can be sequestered in both regions within this range and, thus, these areas of the US agricultural sector are potential participants in a market for C credits.

Influence of program design on costs of C sequestration

The design of policies to sequester soil C will influence the cost of producing or purchasing a given quantity. Abstracting from measurement and monitoring costs, Antle and Mooney (2001), Antle et al. (2001b) and Pautsch et al. (2001) have shown that a per Mg scheme is more efficient than a per hectare scheme, because this design accounts for the spatially heterogeneous ability of land units to sequester soil C and the spatial variation in the opportunity cost of switching production practices. Existing farm support programs, such as the Conservation Reserve Program (CRP) or Environmental Quality Incentive Program (EQUIP), provide payments for each hectare converted to a recommended land management practice. Similarly structured per hectare payments will sequester C less efficiently than a per Mg scheme, since producers are paid a fee that is not directly correlated with the benefits (in terms of the quantity of sequestered C) provided at each location (Antle and Mooney, 2001; Antle et al., 2001b). Stavins (1998, 1999) and Pautsch et al. (2001) suggest that the costs of measurement and monitoring under a per Mg scheme could be prohibitively costly and thus a per hectare payment scheme might be preferred because compliance could be confirmed at a lower cost. However, recent results by Antle et al. (2001b) suggest that the efficiency gains offered by a per Mg program could be large, even accounting for the additional costs of measurement and monitoring.

Costs of Soil C Sequestration in Agriculture versus Alternatives for Mitigating GHG

Antle et al. (2001a) have calculated the marginal costs of supplying C under a range of per hectare payment policies for the Northern Great Plains. The first is a policy under which producers are given a payment if they switch from a crop–fallow system to grass. The second provides producers with a payment if they switch from a crop–fallow system or grass to continuous cropping. Pautsch et al. (2001) also examined a policy that pays producers on an area basis for changing their tillage practices from conventional to no-till.

Table 2 compares the per Mg payment schemes of Antle et al. (2001b) and Pautsch et al. (2001) with the per area payments described above. Results show that some C can be sequestered in US cropland soils for as little as \$10 Mg^{-1} C. At close to \$100 Mg^{-1} C, between 0.3862 and 0.5371 MMgC yr^{-1} can be sequestered in Iowa and between 0.1142 and 0.8536 MMgC yr^{-1} within Montana. Under a per Mg payment scheme both states combined could sequester approximately 1.4 MMgC yr^{-1} at a marginal cost close to \$100 Mg^{-1}.

However, the agricultural sector must also compete with the costs of C sequestered by forest practices and other means. Stavins (1999) and Parks and Hardie (1995) show that C can be sequestered domestically at between \$26 and \$150 Mg^{-1} (Table 3) by changing marginal agricultural lands to forested use. International pilot projects have suggested that C can be sequestered in forest biomass at between \$3 and \$28 Mg^{-1} (Table 3). A comparison of the costs reported in Tables 2 and 3 indicates that soil C sequestration in US cropland is competitive with domestic and international projects; however, the degree of participation and quantity of C supplied (as a proportion of the total traded) will be dependent on the equilibrium market price for C credits.

Discussion and Conclusions

This paper demonstrates that, firstly, economic and biophysical factors need to be considered jointly to assess the efficiency of sequestering soil C and to calculate the size of incentives that

Table 2. Total annual soil carbon sequestration over a marginal cost range of \$0 to approximately \$100 per Mg.

Source	Policy type[1]	Marginal cost (\$ Mg^{1})	Annual carbon increase (million Mg)	Location
Per Mg payment[1]				
Pautsch et al. (2001)	A	13–92	0.0777–0.5371	Iowa
Antle et al. (2001b)	B	10–100	0.3597–0.8086	Montana
Per ha payment				
Pautsch et al. (2001)	C	20–102	0.0777–0.3862	Iowa
Antle et al. (2001a)	D	32–102	0.0197–0.1142	Montana
Antle et al. (2001a)	E	12–98	0.0298–0.8198	Montana

[1]A, producers are paid for each Mg of additional C produced as a result of switching from conventional tillage to no-till.
B, producers are paid for each Mg of additional C produced by a change in cropping system.
C, producers are paid for each hectare that is switched from conventional tillage to no-till.
D, producers are paid for each hectare that is switched from a crop–fallow system to permanent grass.
E, producers are paid for each hectare that is switched from a crop–fallow system or permanent grass to a continuous cropping system.

Table 3. Estimates of greenhouse gas (GHG) emissions reduction costs from US and non US sources.

Source	Marginal cost ($ Mg[1])	Annual carbon increase (million Mg)	Location	Reduction source
Stavins (1999)[1]	0–150	470	US Delta States	Marginal agricultural land changed to forestry us
	0–73	6.35		
Parks and Hardie (1995)[1]	0–26	19.96	US cropland	Marginal agricultural land changed to forestry us
IPCC (2000)	14.25[2]	0.0448	Czech Republic	Forestry
IPCC (2000)	27.80[2]	0.0243	Uganda	Forestry
IPCC (2000)	3.00[2]	0.0114	Belize	Forestry
IPCC (2000)	21[2]	0.0225	Malaysia	Forestry

[1]See Stavins (1999, Table 3) for detailed notes on calculation of annual carbon amount and marginal cost estimates for both studies.
[2]Reported figures are point estimates of the average cost per Mg C to a purchasing agent. This is equivalent to the marginal cost to the producer of producing the last Mg at that quantity.

will encourage producers to participate in a market for C credits. Secondly, the studies compared in this paper demonstrate that the efficiency of soil C sequestration varies spatially in response to the biophysical potential of an area to sequester soil C, and the opportunity cost of management changes. Those areas with the lowest opportunity costs per Mg C will participate in a market for C credits. Under alternative policies, for example a per hectare payment scheme, the spatial distribution of producer participation will differ and favor areas with the lowest opportunity cost per hectare. However, Antle et al. (2001b) show that in the Northern Great Plains, the marginal costs per Mg of soil C sequestered under a per hectare payment mechanism are as much as four times higher than the costs under the per Mg payment mechanism. Policies based on economic or biophysical information alone will result in an inefficient allocation of resources. US policies to encourage C sequestration are likely to be influenced by political as well as economic considerations.

Thirdly, evidence shows that the agricultural sectors within the Northern Great Plains and the Cornbelt could benefit from participation in a market for C credits, both domestically and internationally. There are large areas of the US that are not suited to afforestation and, through soil C sequestration, may be able to participate in a C market. An additional benefit, unlike the conversion of agriculture to forestry uses, is that these land areas will be maintained in production, supporting rural communities.

Fourthly, other important greenhouse gases are omitted from this study but need to be considered in a full accounting of the global warming potential associated with changes in management and production practices. Efficient policies will provide incentives to reduce the total global warming potential of the net changes in emissions and sequestration as a result of changing management practices.

Finally, these results are based on point estimates from each model and, as such, should be treated as only one of a range of possible outcomes. Confidence intervals are not available for either study but are the subject of current research. In addition, these results only consider a subset of potential management changes and although promising, an expanded set may change the opportunities available to producers.

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Incentives for reforestation of agricultural land: What will a market for carbon sequestration credits contribute?

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Abstract. *Reforestation of frequently flooded, marginally productive agricultural lands may provide environmental benefits. However, for reforestation to occur private landowners must earn enough revenues from forestlands to offset both the costs of reforesting as well as any forgone agricultural income. Among the possible revenue sources is the sale of carbon sequestration credits earned from reforestation. We simulate the financial feasibility of reforestation and the contribution of carbon sequestration credit sales to the net income possible from reforestation in a case study area of frequently flooded soybean fields in the Mississippi Alluvial Plain (the Delta). Under current market conditions, reforestation appears to be financially feasible on only the most marginally productive agricultural lands. Markets for carbon credits may contribute to positive financial returns of reforestation, but other policies and market opportunities will be necessary for a positive return to be realized.*

Key words: carbon credit prices, financial feasibility, flooded lands, hunting lease payments, timber revenues, water quality improvement

Introduction

Forest cover on thousands of hectares of flood-prone land in the Mississippi River Alluvial Plain (the Delta) was cleared for soybean (*Glycine max* L.) cultivation during the past century. Reforestation of these frequently flooded agricultural fields may restore the environmental services lost when the land was cleared. For example, reforestation will provide greater cover, nesting sites and brood-rearing habitat for wildlife (Wesley et al., 1981). Reforestation can also improve water quality by reducing the sediment load associated with post-harvest tilling, bedding and residue shedding on soybean fields (Aust et al., 1991; Gianessi et al., 1986) and by reducing runoff of agriculture-related chemicals (Jordan et al., 1993).

Tree species, such as nuttall oak (*Quercus nuttallii*) and eastern cottonwood (*Populus deltoides*) can withstand the frequent and extended periods of flooding that occur in these low-lying areas of the Delta. However, the land is in private ownership and the reforestation will only occur if the landowner's revenue from the reforested land exceeds both the costs of reforesting the land and forgone agricultural income. The sale of timber and other wood products, hunting lease sales and payments from government conservation programs are possible income sources.

The increased international attention to the possibility of climate change through global warming has suggested that carbon (C) sequestration credit sales may also become a source of income from reforested land. Carbon sequestration credit sale opportunities require national or international commitment to reducing net emissions of greenhouse gases (GHG) and policies that allow such commitment to be met, in part, by increasing C sequestration in forest biomass and soils (see http://www.fe.doe.gov/coal_power/sequestration/index.shtml). One policy might be to foster a GHG emissions allowance market, where those who wish to increase emissions of GHG (such as electric utilities) might pay landowners to alter land use in ways that sequester additional C such as reforesting a presently farmed field. An alternative policy might be to make government payments to landowners who reforest for sequestration. In either case, sequestration of C becomes a source of income from reforestation.

This paper reports on the net income that might be earned from the sale of C sequestration credits and examines how such income might contribute to the financial feasibility of reforesting marginally productive agricultural land. The first section of the paper describes the case study area and explains why reforestation of frequently flooded soybean fields in the Mississippi Alluvial Plain may be financially attractive to landowners. The following sections describe a simulation model that was used to calculate the returns to reforestation and the results obtained. These results are then used to draw conclusions about the potential role of C sequestration credit sales in encouraging reforestation in the case study area and elsewhere in the United States.

The Study Area

The Lower Yazoo River Basin in the Mississippi Alluvial Plain ('the Delta') is situated in southern Mississippi. The area is bordered on the east by the Yazoo River and on the west by the

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Mississippi River and the two converge at the southernmost tip of the Basin to create a triangular area of land containing over 218,535 ha (540,000 acres). An estimated 48,968 ha (121,000 acres) of cleared land (primarily agricultural) fall within the 2-year flood plain (50% chance of flooding in any year) and is inundated when high flow on the Mississippi River and its tributaries cause backwater flooding in the spring and early summer (Shabman and Zepp, 2000).

The alluvial soils have a high clay content and poor internal drainage. They are compact, poorly aerated, and quick to saturate and slow to dry when flooded. Thus, crop yields are often low even when flooding is not a deterrent to planting. As a result of the wetness and low yield potential, large areas remained in bottomland hardwood forest until the mid-1970s, when a combination of favorable market conditions and government agricultural policies created incentives for farmers to convert these forests to soybean cropping (Kramer and Shabman, 1993). The cleared fields are now planted to soybean varieties adapted to clay soils and a growing season shortened by the need for the soils to dry out before planting.

These frequently flooded croplands present a set of circumstances where reforestation may be more profitable for the landowner than soybean production. Without the favorable prices of the late 1960s and early 1970s, soybean production on these low-lying fields has proved only marginally profitable. Frequent flooding during the growing season delays planting and damages crops, often requiring farmers to replant or prevents planting altogether (Shabman and Zepp, 2000). Moreover, federal farm programs have not provided significant price or income support to soybean growers, although recent reforms to the federal crop insurance program may result in some subsidies for planting alternative crops in these frequently flooded areas. By reforesting the area, landowners would give up marginal agricultural returns from low-yield soybean production in a market with historically low soybean prices.

Estimating the Net Returns from Reforestation

A Monte-Carlo simulation model was used to calculate the net present value (NPV) of returns per hectare earned from both soybean production and reforestation over a 120-year period. This time period was selected to facilitate the calculation of returns for reforestation regimes with different rotation periods. A rate of 7% was used to discount future annual revenues and costs. All prices and costs are expressed in year 2000 dollars. If there is a positive difference in net returns between reforestation and continued soybean cultivation then reforestation may be financially feasible.

The Monte-Carlo simulation model is designed to account for the effects of flooding on both soybean production and timber stand establishment. Flooding can interfere with tree stand establishment; the more time required to achieve successful establishment, the longer the delay until the trees can be harvested and sold, resulting in more heavily discounted revenues. Similarly, flooding can reduce soybean returns by delaying or preventing planting which will lower production. Also, flooding can destroy a planted crop, requiring the farmer to incur the additional costs of replanting.

The Monte-Carlo simulation model utilizes probability distributions to describe the value of variables affected by the frequency and severity of flooding on a site, such as soybean and timber yields and planting costs. The model repeatedly iterates through the same 120-year period. With each iteration, new variable values are drawn from the probability distributions specified. At the end of the simulation, the net forestry returns calculated for each iteration are averaged to generate a single net present value (NPV) estimate. (For a detailed explanation of the model see Shabman and Zepp, 2000). Specifically, the NPV calculation is:

$$\text{NPV returns from reforestation ha}^{-1} = \sum_{i=1}^{120} [\text{Forest}_i - \text{ReforestCost}_i - \text{Soybean}_i]/(1+r)^i \quad (1)$$

where:

Forest_i = per hectare revenues earned from forested land use in year i;

ReforestCost_i = per hectare costs of reforestation in year i;

Soybean_i = per hectare net soybean returns in year i;

i = year 1 through 120;

$r = 0.07$.

Returns from reforestation

A forested hectare provides a number of services that may be a source of income from reforestation (in Eqn 1, Forest_i). Income from wood product sales, hunting leases, government payments for water quality improvement and from C sequestration credit sales are described in this section.

Timber revenues. Four different reforestation regimes are considered, including cottonwood, cottonwood interplanted with nuttall oak, and two nuttall oak regimes, one using high management practices (referred to as the high management nuttall oak scenario), the second using minimal management practices (referred to as the seeded nuttall oak scenario). Revenues from tree harvests are calculated as the revenues earned from the sale of wood products. Depending on the tree species, revenues earned are based on the sale of pulpwood, timber or both. The present value (PV) of tree harvest returns is calculated by discounting and summing the 120-year stream of timber returns. Table 1, column 1, reports the PV of returns from

Table 1. Present value (PV) ha^{-1} of revenue in dollars from three reforestation income sources.

	Revenue sources		
Reforestation scenarios	Timber	Hunting lease	Water quality payments
Nuttall oak	100.57	244.01	869.00
Nuttall oak (seeded)	85.50	207.42	869.00
Cottonwood	363.19	176.08	869.00
Cottonwood–oak (interplanted)	337.02	238.82	869.00

wood sales for each reforestation scenario. This calculation does not subtract out reforestation costs. As a result, the term 'present value' is used to describe the calculation, rather than the term 'net present value' (NPV). In this paper, the term NPV is used only when costs are subtracted from revenues generated, yielding net returns.

Cottonwood regime. Cottonwood is the fastest growing of the tree species. In the first year, the costs of establishing a cottonwood stand include slitting the ground with a straight shank; hand planting cuttings at a 3.7 m × 3.7 m (12 ft × 12 ft) pattern; multiple applications of herbicide throughout the first year; and two passes over the field with a mechanical cultivator; all at a total cost of $312.20 ha^{-1} ($126.40 acre^{-1}). In the second year, two additional passes are made over the field with a mechanical cultivator, for an additional cost of $14.82 ha^{-1} ($6.00 acre^{-1}). In the simulation model, the cottonwood rotation lasts 10 years. In year 10, the stand is clear cut and sold as pulpwood at $17.37 cord^{-1} (a volume of 3.6 m^3). The pulpwood price is a stumpage price, with the costs of harvesting already subtracted.

If a cottonwood stand is flooded such that the seedlings are overtopped for a duration of at least 2 weeks in the summer months, or at least 4 weeks in the winter months, the model requires that replanting costs be incurred for 80% of the tree stand. For the cottonwood regime, this means a cost of at least $216.80 ha^{-1} ($87.77 acre^{-1}). This assumes that 20% of the tree stand will survive the flooding or will naturally regenerate. It is assumed that the landowner will only attempt to replant once.

Nuttall oak high-management regime. The nuttall oak rotation lasts from 72 to 80 years, depending upon soil type. Establishment of a nuttall oak stand under the high-management regime involves one pass of a bushhog and hand planting of seedlings at 3.7 m × 3.7 m (12 ft × 12 ft) spacing in the first year of the rotation, at a total cost of $267.75 ha^{-1} ($108.44 acre^{-1}). Similar to the cottonwood regime, if the stand of oak seedlings is overtopped by flood waters for more than 2 weeks in the summer months or 4 weeks in the winter, the landowner will incur the additional costs of replanting 80% of the stand, estimated at $173.20 ha^{-1} ($70.12 acre^{-1}). Around years 30 and 55 (actual timing varies slightly according to soil type), the stand is selectively thinned and the trees removed are sold for pulpwood at a stumpage price of $17.37 cord $^{-1}$. At the end of the rotation, the stand is harvested and sold for timber.

The stumpage price of timber at the time of harvest is calculated assuming a 1% annual increase in the current stumpage price of $237.19 MBF^{-1} (thousand board feet; one board foot = 0.00236 m^3 of wood) for the first 10 years of the simulation. Forestry experts at Mississippi State University recommended a 1% annual increase be applied to current timber throughout the life of the tree rotations. This rate of growth is based on recommendations made at a meeting on 22 January 1998, organized by Dr. Steve Bullard at Mississippi State University for the purpose of reviewing and refining the forestry module. However, to allow for uncertainties about future conditions in the wood products market, a more conservative price projection estimate was used. Inflating the current stumpage price by 1% for the first 10 years of the rotation results in a stumpage price for timber of $262.00 MBF^{-1} at the time of final harvest and provides the more conservative estimate used in this model.

Seeded nuttall oak regime. The seeded nuttall oak regime differs from the high-management nuttall oak regime only in the costs of establishing the stand, and in the yields harvested at each of the selective thinnings, and at final harvest. Establishment of the seeded nuttall stand involves only planting the acorns, at a total cost of $136.25 ha^{-1} ($55.16 acre^{-1}). While the costs of establishment are significantly lower, the seeded nuttall oak stand produces only 85% of the yield produced by the high-management stand. The rotation length, timing of selective thinnings and effects of flooding are the same for the two regimes. The same pulpwood and saw timber prices used in the high-management regime were used to calculate revenues earned from the thinning and final harvest for the seeded nuttall regime.

Cottonwood–oak interplant regime. Under the cottonwood–oak interplant regime, nuttall oak seedlings are interplanted at a 50% stocking among a fully stocked cottonwood plantation. This means that, in year 3, 157 nuttall oak seedlings are interplanted among 304 already established cottonwoods. Therefore, the yield at final harvest age for the oaks is estimated to be 50% of the yield computed for the pure nuttall oak plantation, and the cottonwood yields are the same as those calculated for the pure cottonwood plantation. Two cottonwood rotations are completed (10 years each) within the first 20 years. After the second rotation of interplanted cottonwood is harvested, no further interplanting is undertaken and the oak stand is harvested at the end of the nuttall oak rotation. The interplanted cottonwood and oak utilize the same rotation lengths, pulpwood and timber prices, and flooding assumptions that are used in the cottonwood and high-management nuttall oak regimes previously described. An important assumption for the mixed reforestation regimes is that the effects of established cottonwood on oak yields are insignificant (Amacher et al., 1997) This is a reasonable assumption given the established tolerance and slow growth of nuttall oak seedlings (Amacher et al., 1997).

Hunting lease revenues

In the Delta area, leases that allow hunting of multiple species can range anywhere from $3.70 to $61.78 ha^{-1} ($1.50–25 acre^{-1}) while soybean fields and immature timber stands can earn annual hunting lease payments of $12.35 ha^{-1} ($5 acre^{-1}) (Woolfolk 1997). These lease payments change with the tree species and different stages of the rotation. For all tree species, the hunting benefits increase as the tree stand matures. Higher valued sites tend to be mature bottomland hardwood stands, or mature hardwood stands intermixed with agricultural fields, that provide excellent habitat for whitetail deer (*Odocoileus virginanus*), wild turkey (*Meleagris gallopavo)* and rabbit (*Sylvilagus floridanus*). Younger, immature tree stands provide less cover and food for wildlife and, thus, tend to earn lower lease prices. After subtracting out the $12.35 ha^{-1} ($5 acre^{-1}) that could have been earned on the land had it remained in soybean production, the

hunting lease values used in the simulation range from $12.35 to $37.07 ha^{-1} ($5–15 acre $^{-1}$). The PV of hunting lease sales is calculated over the 120-year period for each of the four reforestation scenarios (Table 1).

Government water quality improvement payments

Government subsidy payment programs such as the Wetlands Reserve Program (WRP) may be justifiably used to provide incentives for landowners to reforest, particularly when water quality improvements are expected (CENR, 1999). For the model simulation we calculated a possible government payment amount, if the payment were tied only to the value of the reduced nutrient deliveries from forested versus farmed land. Estimates of the difference between the edge of field nitrogen (N) loads from soybean fields and from forested areas were 11.04 kg ha^{-1} yr^{-1} (9.85 lb acre^{-1} yr^{-1}) (Shabman and Zepp, 2000). Annual N reduction payments were set at $5.53 kg^{-1} ($2.50 lb^{-1}) and were multiplied by the kilograms reduced. The $5.53 kg^{-1} ($2.50 lb^{-1}) payment was an estimate of the cost of nutrient removal at sewage treatment plants (Shabman and Zepp, 2000) under the assumption that these could be avoided if the reduction in N was achieved by reforestation. The annual dollar payment is discounted to present value for each year of the 120-year period. For all tree regimes at all flood frequencies, the PV of N payments over the 120-year period is $869 ha^{-1} (Table 1).

The present value of this nutrient reduction payment is in line with the subsidy payments made for reforestation under the WRP. For example, in Yazoo and Issaquena counties, two of the counties that make up the majority of the land area in the lower Yazoo basin, one-time WRP payments ranged from $494.20 ha^{-1} ($200 acre^{-1}) in 1992 to $1124 ha^{-1} ($455 acre^{-1}) in 1996 (Shabman and Zepp, 2000).

Carbon sequestration credit sales

Carbon is sequestered in three main sinks when agricultural land is converted to forest, i.e., trees, litter and soil. Over time, unharvested trees will sequester greater amounts of C than do agricultural crops that are harvested annually. In addition, conversion of land from agricultural use to forestry typically results in a doubling of soil organic C over the rotation period. Most of this C will remain sequestered until well after the trees are harvested, although the amount depends on site management. (Sampson and Hair, 1996).

The timing and length of sequestration in fields used for forestry is especially important when considering C offsets. To accomplish a reduction in atmospheric C through reforestation, decomposition of biomass and subsequent release of C to the atmosphere must be delayed. In the case of soybeans, which are used as food or fuel, most C is returned to the atmosphere as CO_2 through respiration and combustion. In contrast, woody material used for construction purposes, recycling paper and paper stored in landfills will retain C in an inert state for a considerable period of time. According to one estimate, 60% and 37% of wood-based C is in an inert form 5 and 100 years respectively, following harvest (Row and Phelps, 1990).

The annual rate at which C is fixed in merchantable products is low initially, increases rapidly due to high tree growth rates, and then tends to decrease with the age of the timber stand. Cottonwood, however, maintains high growth rates throughout a 10-year rotation. Table 2 reports estimates of the annual rate of sequestration achieved by hardwood and softwood per hectare for the south-eastern United States for tree species on reforested former agricultural lands. The values were estimated for moderate to good-quality sites under average site management techniques, and show average annual C sequestration rates per hectare at different stand ages (Wainger et al., 2000). The sequestration numbers represent the net increase in C above a baseline scenario in which existing soil C stock was estimated to be 24.80 Mg ha^{-1} (11.0 tons acre^{-1}) with no appreciable annual increment in soil C (based on data from Birdsey, 1996).

The reported numbers represent potential rates of C sequestration, but any activity that results in the release of C sequestered in the tree biomass may reduce the amount of C credits generated by reforestation. For example, these estimates do not account for the release of some of the sequestered C when the trees are harvested. In addition, the estimates reported in Table 2 do not account for the trade-offs that may be required between C sequestration credit generation and managing the stand for water quality, timber or hunting purposes. For example, any timber harvest or selective thinning to improve hunting habitat would require that C credits be adjusted to reflect those activities.

Calculating C revenues. The C sequestration values from Table 2 represent the net accumulation of C above the baseline. These values were used to estimate revenues from the sale of C credits for the four different reforestation regimes considered in this case study. In calculating the C revenues generated by reforestation, it was assumed that if the landowner had chosen to plant the field to soybeans instead of reforesting, credits would have been received for the 24.80 Mg ha^{-1} stock of C associated with the agricultural land. This means that, while the landowner earns C credits by reforesting, he forgoes any revenues earned for the C credits he would have received for cultivating the land in soybeans. Because it was assumed that there is no incremental increase in soil C in subsequent years, the landowner would only earn C revenues for soybean cultivation in the first year of the simulation.

Table 2. Average annual C storage (trees, soil, litter and debris) for reforestation scenarios excluding harvest years.[1]

Age of stand (years)	Planting scenario (Mg ha^{-1} yr^{-1})		
	Nuttall oak	Cottonwood	Cottonwood–oak interplant
0–15	2.35	4.67	5.29
16–50	1.68	4.99	6.13
51–70	1.68	4.84	5.26
51–100	1.88	–	–

[1] Wainger et al. (2000).

For the four different reforestation regimes considered in this case study, the per hectare C revenues earned from reforestation in each year ($CarbonRevenue_i$) are equal to the price of a C credit (one Mg of sequestered C is equivalent to one C credit) multiplied by the total credits of C produced by reforestation that year. The C revenues foregone under soybean production ($AgCarbonRevenues_1$) are equal to the credits earned for the 24.80 Mg ha^{-1} stock of C multiplied by the C credit price. Equation two establishes the calculation used to determine the present value (PV) of revenues from the C credits accumulated over the length of a tree rotation (specific to the tree species planted and the age of the tree stand) with no adjustment for future losses of sequestered C.

$$CarbonRevenue_i = \sum_{1}^{120}[(Price_{Carbon} \times Total\ credits_i)/(1+r)^i] - AgCarbonRevenues_1 \quad (2)$$

where:

$Price_{Carbon}$ = payment received for a C credit ($/credit);

Total credits = Mg sequestered in year i;

$AgCarbonRevenues_1$ = total C revenues that would have been earned in the first year of the simulation for cultivating the field in soybeans.

As with the other reforestation returns, the annual, per hectare C returns were calculated for each of the 120 years included in the simulation. The stream of annual returns was then discounted and summed to provide an estimate of the PV of C credit returns.

Carbon credit prices. Many questions remain about the international programs that will address climate change and GHG limits. As part of the 1997 Kyoto Protocol, the United States committed to reducing greenhouse gas (GHG) emissions by 2008–2012 to 7% below 1990 levels. In the protocol, C sequestering activities were referred to as 'land use, land use changes and forestry' or 'LULUCFs' and could be used as a substitute for emission reductions. However, the US commitment to that protocol has recently been withdrawn. Meanwhile, other nations now have agreed to net GHG reduction targets with a provision allowing those nations to account for C sequestration. However, technical uncertainties remain about how sequestration will be credited. Among the technical issues are the following: (1) assuring that the LULUCF is a newly created means of C sequestration and not just a natural uptake of C that would have occurred independent of human actions, and (2) establishing an accounting system and baselines for determining net amounts of C uptake due to LULUCFs, including determining what happens to C credits when C tied up in tree biomass is released, such as in the event of a forest fire (UNEP, 2000).

We conclude that until potential buyers of credits are certain that sequestration credits purchased today will retain their value under the final international GHG agreements, the price GHG emitters are willing to pay for C credits will be highly uncertain. Therefore C revenues were calculated for different C prices after a review of reports on actual and simulated credit exchanges. One study by the US Council of Economic Advisors simulated prices for an international allowance market in greenhouse gases. The simulated price of \$14 Mg^{-1} of C is found in a July 1998 publication, 'The Kyoto Protocol and the President' Policies to Address Climate Change: Administration Economi Analysis' (CEA, 1998). Natsource, a New York-based energ brokerage firm predicts a trading price in the range of \$6- 40 Mg^{-1} of CO_2 (Pelangio Mines Inc., 2000). Prices as low a \$2.50 Mg^{-1} of CO_2, which is equivalent to \$0.68 Mg^{-1} of C, hav also been suggested (Elias, 2000). We selected the followin prices from this literature: \$0.68, \$6, \$14 and \$20 Mg^{-1} of C.

Results of C revenue calculation. The present value of th revenues from the sale of C credits for each of the fou reforestation scenarios using the four different C credit price selected, are reported as present values (PV) per hectare over th 120-year period of analysis in Table 3. The results represent th sale of credits for only the additional amount of C sequestered b reforestation, above the 24.80 Mg ha^{-1} of C already sequestere by agricultural production. The high-management nuttall oak cottonwood and cottonwood–oak regimes, generated positiv revenues from sale of C credits, ranging from a low of \$1.09 ha for nuttall oak under an assumed price of \$0.68 Mg^{-1} C to a hig of \$924.23 ha^{-1} for cottonwood–oak interplanted under a assumed price of \$20.00 Mg^{-1} C.

In the case of the seeded nuttall oak scenario, however, th revenues from the C credits earned with soybean productio were greater than the C revenues earned with reforestation. Thi result occurs because the revenues earned for C credits under th seeded nuttall oak regime are slowly accumulated during th course of the 120-year planning period, whereas the revenue earned for the 24.80 Mg ha^{-1} stock of C credited to a soybea field are received in the first year of the planning period. Becaus all C revenues earned in future years under the seeded nuttall oa rotation are discounted, the present value of the C revenue under reforestation are less than the undiscounted C revenues fo soybean land that would be earned in the first year of th rotation. As a result, in Table 3, negative revenues ranging fro –\$1.61 to –\$44.31 are reported for C credit sales under th seeded nuttall oak scenario.

For the purposes of this study, it was assumed that, at the tim a nationally recognized C crediting system is established planting soybeans would qualify as a change in land use resulting in a one-time change in the C stock, equal to th 24.80 Mg ha^{-1} of C retained in agricultural soils. The estimate o 24.80 Mg ha^{-1} used in this analysis equals the entire stock of for agricultural land. It is not yet known what baseline will b used for defining changes in the C stock, therefore, th

Table 3. Present value (PV) of revenues ha^{-1} in dollars from th sale of additional C credits generated by reforestation.

	Price Mg^{-1} of sequestered C (\$)			
Reforestation scenarios	**0.68**	**6.00**	**14.00**	**20.0**
Nuttall oak	1.09	11.76	33.85	35.4
Nuttall oak (seeded)	–1.61	–12.33	–23.33	–44.3
Cottonwood	28.42	251.35	582.76	836.7
Cottonwood–oak (interplanted)	31.58	276.50	642.46	924.2

24.80 Mg ha^{-1} was used to establish the upper end of the range of C credits a landowner might receive for a soybean field. If planting soybeans was not considered to be a change in land use for which landowners could earn C credits, then the C revenues reported in Table 3 would increase by the amount of the C price times 24.8 Mg ha^{-1}. A sensitivity analysis was completed to determine whether the calculated NPV from reforestation using these C revenue results would change the overall NPV calculation. While the calculated NPV did increase for each reforestation regime, those regimes with negative NPV remained negative, with the one exception of the cottonwood regime.

Costs of reforestation

The costs of reforestation include both the financial costs of establishing and harvesting a timber stand, as well as the cost of any future agricultural income forgone when the land is reforested. The financial outlays include such expenses as the costs of establishing the forest stand, any costs incurred to replant trees in the event of flood damage, and the costs of hiring a marketing consultant at the time of harvest. The financial costs of reforestation (ReforestCost$_i$ in Eqn 1) are reported in the first column of Table 4.

The agricultural returns calculations consider expected yields, soybean prices, production costs and farm management practices typical of those found in the Mississippi Alluvial Plain. The future agricultural income foregone as a result of reforestation will depend on the yields realized on the land reforested. The two different levels of soybean productivity considered were a flood-free yield of 1.68 Mg ha^{-1} (25 bu acre^{-1}) and a flood-free yield of 2.02 Mg ha^{-1} (30 bu acre^{-1}) (these calculations use 60 lb or 27.24 kg as the weight of a bushel of soybean). These flood-free yields are typical of the conditions in the area; however, even the higher 2.02 Mg ha^{-1} yield is far below the yields realized in most other areas of the US. In addition, flood-free yields for the simulation are adjusted to account for the likelihood and severity of different flooding events in a given year, meaning that expected yields in any year will be below these levels.

Soybean prices are projected for the first 8 years of the simulation using Delta region prices from the Food and Agricultural Policy Research Institute (FAPRI) projections starting in year 2000 and adjusted to year 2000 dollars using macroeconomic projections from USDA Baseline Projections, February 2000 'GDP Price Index'. The projected prices are as follows. Year 1: \$5.31 bu^{-1}; Year 2: \$5.30 bu^{-1}; Year 3: \$5.24 bu^{-1}; Year 4: \$5.13 bu^{-1}; Year 5: \$5.09 bu^{-1}; Year 6: \$5.01 bu^{-1}; Year 7: \$4.95 bu^{-1}; Year 8 through all remaining years of the simulation: \$4.87 bu^{-1} (FAPRI, 1999) (to covert these US values to \$ kg^{-1} multiply the \$ bu^{-1} by 0.03675). Annual soybean returns per hectare were computed using Equation 3. Soybean production costs, prices and yields and flooding effects on expected yield are based on production experiences and the hydrology typical for the Lower Yazoo River Basin (Shabman and Zepp, 2000). Specifically, the forgone agricultural returns are calculated for each year (Soybean$_i$, in Eqn 1) as follows:

$$\text{Soybean Returns} = \text{Price}_{\text{Soybeans}} \times \text{Yield}_{\text{Soybeans}} \times (\text{Reduc} - \text{ProductionCost}_{\text{Soybeans}}) \quad (3)$$

where

Price$_{\text{Soybeans}}$ = price per bushel of soybeans;

Yield$_{\text{Soybeans}}$ = flood-free soybean yield per hectare;

Reduc = yield reduction factor applied to annual flood-free yield to account for effect of flooding;

ProductionCost$_{\text{Soybeans}}$ = annual soybean production costs.

Estimates of the net soybean returns for both yield assumptions are reported in Table 4.

Net Returns to Reforestation

In this section we report the calculated net returns for different combinations of forestry revenue sources. First we consider the net returns from only timber and hunting lease sales, since these are the most conventional goods and services of reforestation. Then, we consider government payments for water quality improvement and the contribution of C sequestration payments. In this analysis we consider the NPV for both the lowest yield and higher-yield agricultural lands.

Net returns with timber and hunting lease sales

The returns to reforestation net of the financial and opportunity costs of reforesting and including only revenues from the sale of wood products and hunting lease sales are reported in Tables 5 and 6. On the low-yield lands (1.68 Mg soybean ha^{-1}), the returns from timber are positive for the cottonwood and cottonwood–oak regimes. For the high-manage-

Table 4. Present value (PV) in dollars of reforestation costs and soybean returns foregone with reforestation.

Reforestation scenarios	Reforestation costs	Forgone soybean returns (yields ha^{-1}) 1.68 Mg	2.02 Mg
Nuttall oak	301.81	271.12	1200.36
Nuttall oak (seeded)	127.97	271.12	1200.36
Cottonwood	450.61	271.12	1200.36
Cottonwood–oak (interplanted)	519.43	271.12	1200.36

Table 5. Net present value (NPV) of timber returns for 1.68 Mg ha^{-1} flood-free soybean yields and present value (PV) of hunting lease sales (dollars).

Reforestation scenarios	NPV of timber returns	Hunting lease	Total
Nuttall oak	–170.55	244.01	73.46
Nuttall oak (seeded)	–185.62	207.42	21.80
Cottonwood	92.07	176.08	268.15
Cottonwood–oak (interplanted)	65.90	238.82	304.72

ment and seeded nuttall oak regimes, however, the timber returns alone are not adequate to offset the income from continued farming. However, if hunting lease sales can be secured, then the returns from reforestation will be positive. Returns from the cottonwood–oak interplant are the highest. It follows that the addition of other revenue sources for reforestation on the lower-yielding lands only serves to make the reforestation option more attractive to landowners.

On the higher-yielding lands, no combination of timber and hunting lease sales possibilities makes reforestation financially positive. What we will now consider is whether additional income sources will turn the financial feasibility of reforestation positive on the more productive agricultural lands.

Net returns with all revenue sources

The results in Table 6 indicated that, on the higher yielding agricultural lands (2.02 Mg soybean ha^{-1}), the NPV of reforestation for all timber regimes is negative, or only marginally positive, when timber and hunting lease sales are the only revenue sources. Table 7 reports the improvement in the NPV o reforestation if C credit sales are added.

Table 6. Net present value (NPV) of timber returns assuming 2.02 Mg ha^{-1} flood-free soybean yields and present value (PV) of hunting lease sales (dollars).

Reforestation scenarios	NPV of timber returns	Hunting lease	Total
Nuttall oak	–1099.8	244.01	–855.78
Nuttall oak (seeded)	–1114.9	207.42	–907.44
Cottonwood	–837.17	176.08	–661.09
Cottonwood–oak (interplanted)	–863.34	238.82	–624.52

The result for NPV if C credit sales are added to hunting leas and timber sales is reported in column 3, Table 7. The additiona C revenues make the difference between a profitable an unprofitable forestry enterprise on higher-yielding lands, in som cases. For example, at a C credit price of \$20 Mg^{-1}, th cottonwood NPV changes from –\$661.09 to \$175.66 ha^{-1}. The cottonwood–oak regime increases from –\$624.52 to \$17.94 ha^{-1} at a C credit price of \$14 Mg^{-1} and from –\$624.52 to \$299.7 ha^{-1} at a C credit price of \$20 Mg^{-1}. For the most part, however the NPV of reforestation remains negative, or only marginally positive, with the addition of C credit sales.

These results suggest that some form of government payment may be necessary to provide sufficient incentive to promp reforestation of privately owned agricultural lands. One example is the calculated \$869 ha^{-1} payment for reduced nutrien deliveries reported in Table 1. When added to hunting lease timber and C returns, this additional government payment woul be sufficient to make the NPV of reforestation on higher-yielding agricultural lands positive for all regimes, except for the seeded nuttal oak regime.

Conclusions and Implications

While reforestation of marginally productive farmland may be environmentally beneficial, reforestation only will occur i landowners find the revenue from reforestation to be greate than revenue for continued soybean production. The result reported in this paper suggest that income from timber an hunting leases alone makes the NPV of reforestation positive o

Table 7. Net present value (NPV) of reforestation returns ha^{-1} without and with C revenues in dollars, assuming flood-free soybean yields of 2.02 Mg ha^{-1}.

Reforestation scenarios and C credit prices (\$ Mg^{-1})	NPV reforestation returns, excluding C revenues	NPV reforestation returns including C revenues
Nuttall oak		
0.68	–855.79	–854.70
6	–855.79	–844.03
14	–855.79	–821.94
20	–855.79	–820.36
Nuttall oak (seeded)		
0.68	–907.48	–909.09
6	–907.48	–919.81
14	–907.48	–930.81
20	–907.48	–951.79
Cottonwood		
0.68	–661.09	–632.67
6	–661.09	–409.74
14	–661.09	–78.33
20	–661.09	175.66
Cottonwood–oak (interplanted)		
0.68	–624.52	–592.94
6	–624.52	–348.02
14	–624.52	17.94
20	–624.52	299.71

the most marginally productive lands. Recall that these lands had a 50% chance of flooding in any year (the 2-year flood plain) and had soils that produced yields under flood-free conditions of about 1.69 Mg ha^{-1}. Such land would be among the least productive in all of the US. However, only a small improvement in the yield of the flood-prone lands (the land would still have a 50% chance of flooding) would make the calculated NPV for reforestation negative, if just the income from timber and hunting leases were available to the landowner. Even with the addition of C payments, the NPV for reforestation remains negative, or only marginally positive. The results of the simulation model suggest that on these slightly more productive lands all income sources, including timber, hunting lease and C credit sales as well as payments through a government conservation program, would be required for the NPV of reforestation to be positive. These results suggest that programs and polices to encourage reforestation should be targeted to only the most marginally productive lands. However, it is likely these marginally productive lands (frequently flooded, heavy soils and often in the riparian zone) when reforested will offer significant environmental benefits in a watershed.

One possible conclusion from these results is that neither government payments nor C credit sales will be necessary for encouraging reforestation of the most marginal lands. In addition, it might be concluded that existing timber and hunting lease markets are sufficient for encouraging reforestation. Such conclusions would be mistaken, because the measure of financial feasibility, NPV, simulated for this paper may not be the decision criterion used by a private landowner. For NPV to be the reforestation decision criterion the landowner needs to have a long time horizon and be fairly certain about the future income opportunities from the reforestation investment. However, the income flow from timber sales occurs in large increments in a limited number of years over the 120-year simulation period. The NPV calculation discounts the income from the distant future, but does not account for the uneven flow of that income. Such an uneven income stream might not be attractive to landowners, especially given the uncertainty of future timber yields and prices over a 120-year time horizon. Hunting lease sales are strong at present, but the market price for leases could well be affected as the supply of leases increases. Therefore, the financial attractiveness of reforestation to owners of even the more marginally productive lands may be misrepresented by NPV results that include only future, and perhaps uncertain, timber product and hunting lease sales. Some additional government and non-governmental policies and programs may be needed.

One solution to the cash flow unevenness (and uncertainty) of forest product sales is to encourage intermediaries, perhaps analogous to farm marketing cooperatives, to become the responsible party for the reforestation. These intermediaries would have a longer time horizon than the landowner and so would find the positive NPV attractive. They could contract with landowners and offer an up-front cash payment that is the expected value of the timber at a future date. In return, the intermediaries assume responsibility for timber management on the land and receive the rights to the harvest. For example, a Nature Conservancy program in the Clinch Valley of Virginia has exactly this relationship with landowners (Nature Conservancy, 1998). Forest product firms may also serve this role, given their longer business time horizon.

Certainly, C sequestration sales opportunities will await a more certain international commitment to GHG gas reduction and clarification of the role of sequestration in meeting reduction targets. However, if this sales possibility develops, the same intermediaries who might contract for timber harvest rights, as discussed above, might also become the brokers for the C sequestration credits. The intermediaries would assume the responsibility for the retention and management of the forest cover and would keep account of the sequestered C. A share of the C payments would be transferred to the landowner who sold the harvest rights to the intermediary and could be negotiated as part of the contract between the landowner and the intermediary.

The existing demand for hunting lease opportunities could be expanded. Several factors can influence the price of hunting leases and should be considered when marketing leases. For example, landowners who are nearer to urban centers have a greater potential for the sale of hunting leases. Also, for the landowner to be successful in leasing land, a substantial area of reforestation may be required, especially for deer habitat. This means that if neighboring landowners also reforest, then there is increased chance of securing higher price for hunting leases. However, if these same landowners also sell hunting leases, then the additional supply of leases may in fact drive prices down. Market development in this context may mean three actions. First, technical assistance and financial planning advice may be offered to landowners who are seeking to sell leases. Secondly, a program to match buyers of leases with sellers, even through an Internet site, could be considered. Thirdly, any advertising or other programs to increase demand for leases in the face of increasing supply could help to maintain lease prices over time.

These are all possible means for making the income possibilities from reforestation more certain and more immediate for the landowner. Alternatively, reforestation would be encouraged if the landowners had access to an up-front payment, or a predictable annual payment, from government. This is especially true given that hunting lease markets are still developing, that forest product (and C sales) intermediates may not be in place, and the still significant uncertainties concerning the future prospects for sale of C sequestration credits. The challenge for the future will be to develop and then coordinate polices that provide these alternative income opportunities.

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Carbon offsets for conservation and development in Indonesia?

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Abstract. *The logic of the Clean Development Mechanism (CDM) and other such 'carbon (C) offsets' rests on the notion that the opportunity cost of reducing C emissions is lower in developing countries, creating opportunities for mutually beneficial C trading. While the CDM may offer significant gross financial benefits, there has been little analysis of the opportunity costs of foregone resource exploitation and development opportunities. In addition to assessing the potential for net benefits, this paper also considers the practical implications for design and implementation of C offsets for forest conservation and agroforestation in Indonesia. Virtually nothing is known about these administrative factors, and their associated costs, which can also play a decisive role in feasibility of C offset schemes. Data are from field studies in the lowlands of the Indonesian island of Sumatra, where forests and derived land uses are broadly representative of the lowland humid tropical rainforest systems of insular South-East Asia. These data are the basis for assessing the trade-offs between natural forest protection for C sequestration and conversion to other land uses to meet national development objectives that directly affect people's livelihoods. Carbon stocks are analyzed in terms of 'time-averaged' C, an indicator developed for this project. Opportunity costs of land-use alternatives are estimated using standard techniques for economic assessment of investment projects in developing countries. The study finds that imputed timber values are a significant share of the opportunity costs of forest conservation, even for conservative estimates of timber prices. The question of compensating for these foregone values raises complex questions regarding the political economy of Indonesia, since property rights over these resources are highly contentious. These controversies over property rights are also examined within the context of recurrent smoke pollution from land fires in Indonesia, that (among other problems) contribute to greenhouse gas emissions. This smoke is symptomatic of deeper political and institutional problems that raise questions about the permanence of C storage through land-use practices in Indonesia. In comparison to forest conservation, C offsets through agroforestation seem more feasible in Indonesia because property rights over timber from planted trees would be easier to establish and enforce than property rights over timber from natural forests. Although results in this paper look promising, there is still much to be learned about implementation costs, even for the 'easier' case of agroforestation.*

Key words: agroforestry, carbon sequestration, Clean Development Mechanism, Kyoto Protocol, policy reform, sustainable development, tropical forests

Introduction

Per capita carbon (C) emissions are much higher in the United States and other rich countries of the Organization for Economic Cooperation and Development (OECD) than they are in lower-income 'developing' countries. This is largely because of differences in consumption of fossil fuels, that (with cement manufacture) account for roughly 80% of C emissions. Although the link is not technologically predetermined, there is a strong positive relationship between C emissions and income per capita (WRI et al., 1998, p. 344–345). The logic of the proposed Clean Development Mechanism (CDM) in Article 12 of the 'Kyoto Protocol' to the UN Framework Convention on Climate Change (UNFCCC) rests on the notion that the cost of reducing C emissions is lower in developing countries, creating (at least in principle) opportunities for mutually beneficial C trading. Beyond opportunities for trading of emissions in the energy and industrial sectors, the Kyoto Protocol also raises the general possibility of offsetting C emissions with C 'sinks' in land use, land-use change and forestry (LULUCF). It is not yet resolved whether (much less how) LULUCF activities are included in the CDM.

There are clear benefits for the US in such arrangements. In a recent survey, Cooper (2000, p. 165) reported that the marginal costs of meeting Kyoto targets for the US would fall from $200 per Mg C without emissions trading, to $56 per Mg from trade among developed and transitional economies. If trading with some developing countries (China and India) were allowed, the

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marginal cost of C emission abatement could fall below $25 per Mg.

Would payments for C sinks—at say $25 per Mg—be a good proposition for the people of Indonesia and other tropical developing countries? Some have argued idealistically that 'artificial' distinctions between global environmental interests and regional, national and local concerns impede action (UNDP et al., 1994, p. 5), but the trade-offs among objectives spanning these scales cannot be ignored by negotiators or by people directly involved. Results of the Alternatives to Slash-and-Burn (ASB) program, which is introduced below, indicate that pursuing global environmental interests sometimes involve a high opportunity cost for local people in Indonesia. However, there may also be some scope for finding common ground to couple local development initiatives with global interests in C sequestration, since, if the possibility of global climate change is realized, its local manifestation may accentuate the frequency and scale of floods, droughts, fires and pest outbreaks (Jepma and Munasinghe, 1998, p. 49). Indeed, quite apart from C trading, the UNFCCC recognizes the 'adaptation' costs developing countries face if likely climate change scenarios are realized. The need for adaptation, of course, feeds back to the purpose of the CDM, which explicitly aims at sustainable development (without defining what 'sustainable' means, or even indicating who decides on the criteria of sustainability).

The benefits of CDM-type arrangements for C sinks in Indonesia and other tropical countries may yet turn out to be a moot question. No OECD country had ratified the Kyoto Protocol at the time of this writing (January 2001). Prospects for agreement on the CDM seemed particularly poor, in no small part because of unresolved ethical and equity issues (Baer et al., 2000). And the particular question of including C 'sinks' in the CDM is especially contentious. However, it is not clear whether the failure of negotiations in the Hague in November 2000 signals the end of a process spanning a few years or the beginning of one taking decades. Henry Jacoby, paraphrased in Science (Kerr, 2000, p. 921), noted 'that it took 50 years for the General Agreement on Tariffs and Trade to evolve into the 138–nation World Trade Organization. Kyoto might evolve the same way'; and Morgan (2000) has argued that Kyoto is not the only way to proceed, indeed it may not even be the best way ahead, 'The history of international environmental protection shows that effective regimes start slowly. The diplomatic community should work to encourage the growth of local and regional regimes . . . so that they can ultimately coalesce into a comprehensive set of global arrangements'.

With all the uncertainty, it is not surprising that there has been little progress on questions regarding implementation of the CDM raised by Sedjo et al. (1998) and others soon after the Kyoto Protocol was put forward. This paper reports on aspects of ASB research on two types of land use change in Indonesia—deforestation and agroforestation—to identify priorities for action research on these implementation issues. Although there are gaps in our knowledge of technological options and measurement methods, the institutional and policy questions are even more challenging. Regardless of the fate of Kyoto, there would appear to be a role for the international community to support Morgan's (2000) call for efforts to 'speed adaptive learning based on a sharing of experience of early adopters'. But, as will be discussed in the concluding section, whether or not it is in Indonesia's interest to participate in these experiments in institutional innovation depends as much (or more) on the strategies adopted by private firms in OECD countries as on the policies of their governments.

Carbon Offsets for Forest Conservation

In line with the spirit of the sustainability clause underlying CDM, the goals of the global Alternatives to Slash-and-Burn (ASB) program are to identify means to reduce the rate of tropical deforestation driven by slash-and-burn and to reduce poverty of smallholders dwelling at the forest margins. ASB was formulated as a partnership among national and international institutions to undertake research on sustainable upland systems as alternatives to unsustainable slash-and-burn in various parts of the tropics. Measurement of differences among environmental consequences of the various land uses provides the basis for quantifying major trade-offs involved in land-use change. Among the main goals of ASB was assessment of the implications of alternative land-use practices on 'climate change' drivers (C sequestration and greenhouse gas fluxes) and on sustainable development (including growth, poverty alleviation and food security, as well as agronomic indicators of sustainability). Results of these studies, and the new tools developed by ASB to obtain the necessary data, are reported in separate documents (Palm et al., 1998; Tomich et al., 1998a, 1998b, 2001; Vosti et al., 2000).

The island of Sumatra was chosen to represent the lowland humid tropical forest zone in Asia for the global ASB project. Within Sumatra, ASB research to date has focused on two of the five major agro-ecological zones: the broad peneplain zone and the narrow piedmont (foothill) zone (Scholz, 1983; van Noordwijk et al., 1995). The peneplains in particular have been the focus of government-sponsored settlement schemes (called transmigration), road construction, large-scale logging and various large-scale public- and private-land development projects since the 1970s. Because of these activities, little natural forest remains in Sumatra's peneplains. This process of deforestation, which is almost complete in lowland Sumatra, seems likely to be repeated elsewhere in Indonesia. By understanding this process and its consequences in Sumatra, ASB researchers hope to identify policies and technologies that can ameliorate the effects of deforestation and contribute to conservation of the remaining rainforests in Asia.

Sumatra's peneplains are home to millions of small-scale farmers—including indigenous groups, spontaneous migrants and government-sponsored transmigrants—all of whom depend primarily on land converted from forest in order to make a living. Significant numbers also gather products from the forest. Public and private estates (operating forest concessions and plantations of 10,000–300,000 ha or more) pursue profitable large-scale land uses in the same areas. Large estates and smallholders compete for a limited area of land, which

contributes pressure for forest conversion. The land uses and management strategies of large-scale estates differ significantly from smallholders' land-use alternatives in their social, economic and environmental impacts.

Sumatran land uses

Seven 'meta' land uses were selected to organize the national ASB research agendas in a way that would facilitate cross-site comparisons (Table 1). Because deforestation is among the primary concerns of this research, natural forests provide the basic reference point for global environmental concerns. Grasslands and pastures are included as reference points at the opposite ecological extreme. In between, a representative range of five generic upland, rain-fed land-use systems were selected for cross-continent comparisons of alternatives: (1) extraction of forest products; (2) complex multistrata agroforestry systems, also known as 'agroforests'; (3) simple treecrop systems, including, but not limited to, monoculture; (4) crop–fallow systems, which include the textbook version of 'shifting cultivation' or slash-and-burn agriculture; and (5) continuous annual cropping systems, which may be monocultures or mixed cropping. More detailed definitions of these six land uses can be found in Tomich et al. (1998a).

Carbon stocks

Carbon stocks in above-ground vegetation only are included in this analysis. Carbon in the roots and the topsoil (0–20 cm) were also measured but were excluded because of the high variability in those data and the relatively small changes in these stocks compared to those of above-ground carbon (Palm et al., 1998). (The systems studied all are on mineral soils; this generalization does not apply to peat soils.) Greenhouse gas emissions (methane and nitrous oxide) were also measured for the same land-use systems as studied for their C stocks. However, these fluxes from living systems are a tiny fraction of the C release resulting from forest conversion (Tomich et al., 1998a, Part II). The point data from the samples of above-ground C stocks were used to estimate the 'time-averaged C stock' for major land-use systems (Palm et al., 1998). Land use/cover change can thus be translated into a net release or net storage of C.

C stocks of tree-based land-use systems depend largely on the typical cycle length of these systems, as annual C increments are similar. Thus, time-averaged C stocks are similar for forest extraction and long-rotation tree-based systems, such as agroforests (Table 2). However, it must also be emphasized that agroforests and other tree-based systems are not perfect substitutes for biodiversity conservation in natural forests. Indeed, conversion of natural forests to agroforests involves a significant reduction in species richness (Michon and de Foresta, 1995; Tomich et al., 1998a).

Profitability of forest-derived land uses

ASB methods for profitability estimates apply the 'policy analysis matrix' approach of Monke and Pearson (1989). Each is the present discounted value of estimates of cash flows over 25 years. A real discount rate of 15% per year was used for the social profitability estimates reported in this paper. Other details are given in Tomich et al. (1998a) and Vosti et al. (2000). Although efforts were made to adjust for trade policy distortions, it was not possible to account for environmental externalities in these social profitability estimates.

At the time of the ASB field studies (in mid-1997) in Sumatra, estimates of returns to land indicate that the textbook 'shifting cultivation' system—upland rice (*Oryza sativa* L.) followed by bush fallow—was no longer profitable (Table 2). This is consistent with the disappearance of shifting cultivation in most of Sumatra's peneplains. Sustainable forms of continuous foodcrop production may be technically feasible in Sumatra's peneplains, but often are not financially attractive because they require too much labor and too many purchased inputs, as in the case of cassava (*Manihot esculenta*).

The main land uses of the peneplains—smallholders' rubber (*Hevea brasiliensis*) agroforests and large-scale oil palm (*Elaeis guineensis*) plantations and commercial logging—offer higher returns to land than community-based forest management (Table 2). However, these estimates are a lower bound for profitability of community-based forest management, for at least two reasons. First, it was not possible to cover all the myriad commodities collected from the forest by local villagers. Second, because restrictions banning logging by villagers are enforced actively, it was not possible to obtain data about villagers' timber extraction from this forest.

The results for commercial logging also require some explanation. The negative US$13 ha^{-1} (all values are in year 2001 dollars) for commercial logging may appear paradoxical, but this is because sustainable logging regulations—if they really are followed—reduce profitability, mainly by slowing timber extraction. However, timber companies can get around these regulatory 'problems'. First, many companies circumvent regulations on timber extraction. Second, these are typically vertically integrated firms, producing products such as plywood for the export market. The best indicator of profitability of these activities by logging companies is the figure of just over US$875 ha^{-1} in Table 2, valued at prices that reflect world prices of forestry products. (Tomich et al., 2001). Although this profitable form of logging probably is not a sustainable form of forest management, logging can (and often is) part of the process of conversion to non-forest land uses that are sustainable. The figure reported here is a lower bound for the profits from clear felling to clear land, and the profits from logging could also be added to the profitability estimates for land uses derived from natural forests, particularly oil palm monoculture, examined below.

Trade-offs between carbon stocks and potential profitability

Compared to natural forests, both of the forest extraction activities and all the forest-derived land uses studied reduce C stocks (Table 2). These range from about 30–40% reductions for extractive activities (community-based forest management and

Table 1. Major land uses at the forest margins of the peneplains of Sumatra, Indonesia.[1]

'Meta' land use	Corresponding land use in lowland Sumatra	Type/scale of operation	Landscape mosaic context	Description
Natural forest	Natural forest	25–ha fragment within a logging concession	Forest mosaic	Reference point: primary baseline for assessment of land-use alternatives. Undisturbed for at least 100 yr
Forest extraction	Community-based forest management	Common forest land of 10,000–35,000 ha	Indigenous smallholder landscape mosaic	Reference point/possible ASB best bet: products are honey (every 3 years), fish, petai
				Reference point/possible ASB best bet: products are honey (every 2 years), fish, petai, rattan and songbirds
	Commercial logging	Logging concession of 35,000 ha or more	Forest mosaic	Reference point/best bet from official perspective: simulation of Indonesian 'sustainable logging system'; 40-yr cycle
				Reference point: based on estimates of actual harvesting behavior for a concession that recently has been renewed; 20–25-yr cycle
Complex, multistrata agroforestry systems	Rubber agroforests	Smallholders' plots of 1–5 ha	Indigenous smallholder landscape mosaic	Indigenous system: forest clearing followed by upland rice and planting of 'unselected' rubber seedlings, with natural regeneration of forest species. This is the dominant smallholder land use
	Rubber agroforests with improved planting material	Smallholders' plots of 1–5 ha	Indigenous smallholder landscape mosaic	Possible ASB best bet: forest clearing followed by upland rice and planting of rubber clones, with natural regeneration of natural forest species
Simple treecrop systems	Rubber monoculture	Smallholders' plots of 1–5 ha	Indigenous smallholder landscape mosaic	(Formerly) best bet from official perspective: upland rice and planting of rubber clones, with intensive use of inputs and labor to prevent regeneration of natural forest species
	Oil palm monoculture	Large-scale private estate of 35,000 ha or more	Monoculture plantation	Best bet from official perspective: plantation oil palm grown in close association with processing mill (processing not included in the economic analysis)
Crop/fallow systems	Upland rice/bush fallow rotation (shifting cultivation)	Smallholders' plots of 1–2 ha yr^{-1}, often located in community land	Indigenous smallholder landscape mosaic	Reference point: 1 year of upland rice followed by bush fallow of 10 years of more. The dominant smallholder land use of 100 years ago, now rare
				Reference point: 1 year of upland rice followed by a short bush fallow of 5 years or less. Now found only in isolated areas
Continuous annual crops/ grasslands	Continuous cassava degrading to *Imperata cylindrica* grassland	Smallholders' plots of 1–2 ha within large-scale settlement project	Large transmigration project divided into small plots	Reference point: monocrop cassava with little use of purchased inputs
				Reference point: monocrop cassava with intensive use of purchased inputs

[1] Source: Tomich et al. (1998a).

commercial logging) to 85% reduction for continuous foodcrops degrading to *Imperata cylindrica* grasslands. (We explore possibilities for agroforestation of C-depleted *Imperata* grasslands below.)

Intensified tree-based systems—rubber agroforests planted with improved clones or hybrid oil palm monoculture—combine attractive returns to land with 35–40% of the time-averaged C stocks of natural forests. Although the profits for conversion of natural forests to these land-use systems are high (especially if timber sales from land clearing are added to the returns to the land use), the possible values of C sequestration services are even higher. Profitability estimates in Tables 3 and 4 are converted from Indonesian Rupiah to US$ at pre-monetary crisis exchange rate of Rp 2400 per US$ of mid-1997, which arguably better reflects a long-run equilibrium exchange rate than the drastically devalued rates that have prevailed during the monetary crisis and ensuing political turmoil.

For natural forest conservation—the 'pristine' forests that are the baseline for ASB analysis—Table 3 presents estimates of C stocks saved and opportunity costs in terms of land-use alternatives, with and without including the value of logging. Opportunity cost in this table means the net present value of foregone returns to the alternative land use. The change in C stocks in Table 3 is the difference between the above-ground C

Table 2. Carbon stocks and potential profitability.[1]

Land use	Scale of operation	Carbon stocks (above ground): time averaged (Mg ha^{-1})	Potential profitability: returns to land at social prices[2] (US$ ha^{-1})	Employment: time-averaged labor input (days ha^{-1} yr^{-1})
Natural forest	25-ha fragment	254	0	0
Community-based forest management	35,000 ha common forest	176	3–7.5	0.2–0.4
Commercial logging	35,000-ha concession	150	(13) to 875	31
Rubber agroforest	1–5-ha plots	116	30	111
Rubber agroforest with/clonal planting material	1–5-ha plots	103	97–1510	150
Oil palm monoculture	35,000–ha estate	91	617	108
Upland rice/bush fallow rotation	1–2-ha plots	74	(75)	15–25
Continuous cassava degrading to *Imperata*	1–2-ha plots in a settlement project	39	(131)	98–104

[1] Source: adapted from Tomich et al. (2001), Table 12.2, p. 230–231.
[2] At the time of the study in 1997, the conversion rate was Rupiah 2400 per US$1.
Note: numbers in parentheses are negative.

Table 3. Carbon emissions avoided and opportunity cost of natural forest conservation.[1]

Land use averted	Carbon stocks saved (above ground): change in C time averaged (Mg ha^{-1})	Opportunity cost of land-use averted: change in returns to land[2] (US$ ha^{-1} at social prices)	Payment per Mg C to offset opportunity cost (US$ Mg^{-1} C)	Maximum feasible transaction costs for world price of US$25 Mg^{-1} C (%)
Community-based forest management	78	8	0.10	>99
...with logging	104	883	8.49	66
Rubber agroforest	138	30	0.22	>99
...with logging	138	906	6.57	74
Rubber agroforest with/ clonal planting material	151	98–1509	0.65–9.99	60–97
...with logging	151	up to 2385	up to 15.79	as low as 37
Oil palm monoculture	163	617	3.78	85
...with logging	163	1493	9.16	63

[1] Source: calculated from Table 2.
[2] Currency conversion at Rp 2400/US$, the rate prevailing in mid-1997 at the time of these surveys.

stocks of natural forests and the time-averaged C stocks of the alternative land use.

Based on these estimates of C stocks saved through forest conservation and the opportunity costs of land uses averted, Table 3 also presents estimates of the farmgate (or forestgate) payment per Mg of C needed to offset incentives for forest clearing (simply estimated opportunity cost per hectare divided by the estimated Mg of C saved per hectare). Putting aside timber values for a moment, the range of values of C payments necessary to shift incentives from conversion to conservation varies from $0.10 per Mg for community-based forest management to under $4 per Mg for large-scale oil palm plantations to $10 per Mg for rubber agroforests if much of the productivity growth potential 'on the shelf' can be realized.

Indonesia's lowland dipterocarp forests are a very valuable timber resource, which is readily marketable. Thus, it is not surprising that adding even a lower-bound estimate for the value of logging (the 'with logging' rows in Table 3) has a significant effect on payments per Mg of C saved necessary to offset conversion incentives. With a rough proxy for timber included, the range of estimates increase from $0.10 to almost $8.50 for community-based forest management, to almost $10 for oil palm, and to nearly $16 per Mg for intensified rubber agroforests.

The estimates in Table 3 suggest that a world price of $25 per Mg of C could shift incentives from forest conversion to conservation, if these payments reach the people making the decisions and agreements are enforceable. As will be discussed further below, it is not clear how such transactions would work, and little is known about how much these 'costs of doing business' (transaction costs) would be. The final column of Table 3—which is the difference (in percent) between a notional world price of $25 per Mg of C and the estimated payment necessary per Mg—gives some idea of the maximum transaction cost that would still provide a sufficient incentive to local land users. Although the opportunity costs are high, the very large C increments from saving natural forest bring down the necessary payment per Mg of C and leave a margin of 37% or more for transaction costs.

However, there is little 'pristine' natural forest left to save in lowland Sumatra. Table 4 presents estimates for a more realistic scenario: conservation of community-managed forest. Compared to natural forest, these are partially depleted of C but they also involve some limited livelihood opportunities. The C stocks saved in Table 4 are the difference between the lower measured values for community-managed forests and the land use averted. Similarly, the opportunity cost column in Table 4 is the difference between the higher estimate of returns to community-based management ($8 per ha) and the estimate of the present value of the option of converting to rubber agroforest (for smallholders) and oil palm plantation (for large-scale plantation companies). The lower, probably more realistic increment in C saved in Table 4 (compared to Table 3), results in much higher payment per Mg of C required to offset opportunity costs and correspondingly reduces the margin available to cover transaction costs. When timber values are included (the 'with logging' rows), the direct payments required per Mg C rise to about $15–20 for smallholders' conventional agroforests and oil palm plantations.

Most of the calculations in Tables 2, 3 and 4 are based on technologies that predominate currently in Sumatra. However, the possibility of significant yield increases for rubber agroforests exists through adoption and adaptation by smallholders of higher-yielding rubber clones which are on the 'shelf' of available technologies. Field experience still is limited, but prospects are good that profitability of planting clones in smallholders' agroforests exceeds profitability of planting seedlings (the present norm). The question is, by how much could profitability be increased and over what time period? Certainly, within the time frame of C offset projects, there is

Table 4. Carbon emissions avoided and opportunity cost of community-based forest management.[1]

Land use averted	C stocks saved (above ground): change in C time averaged (Mg ha^{-1})	Opportunity cost of land use averted: change in returns to land[2] (US$ ha^{-1} at social prices)	Payment per Mg C to offset opportunity cost (US$ Mg^{-1} C)	Maximum feasible transaction costs for world price of US$25 Mg^{-1} C (%)
Rubber agroforest	60	23	0.38	98
...with logging	60	899	14.98	40
Rubber agroforest with/ clonal planting material	73	90–1502	1.23–20.57	18–95
...with logging	73	up to 2378	up to 32.57	not feasible
Oil palm monoculture	85	609	7.17	71
...with logging	85	1485	17.47	30

[1] Source: calculated from Table 2.

[2] Currency conversion at Rp 2400/US$, the rate prevailing in mid-1997 at the time of these surveys.

significant scope for technological change. Results in Table 4 indicate that if available rubber clones were adopted and their yield potential realized by smallholders in rubber agroforests, then the payment required to offset the opportunity cost of this land use could exceed $30 per Mg C.

Knowledge gaps regarding feasibility of C offsets through forest conservation

Smith and Scherr (2001) observe that 'most pilot forest protection projects have been carried out in remote areas with low population densities, where the rate of deforestation in the absence of the project is moderate or low'. This raises questions about the 'additionality' of the C stocks saved—perhaps they were not at great risk. It also provides a probable explanation why most such projects are in the sparsely populated areas of the Amazon. In densely populated South-East Asia, establishing that natural forests (and hence their C stocks) are at risk is straightforward, but this also comes with higher opportunity costs of alternative land uses and (probably) higher transaction costs of protecting forests.

Estimates in Watson et al. (2000) indicate that many forest protection projects may be able to supply C at under $5 per Mg. This is consistent with the results in Table 3, as long as values from logging of timber are ignored. However, even very conservative estimates for timber values for lowland Sumatra raise estimates of necessary payments well above $5 per Mg of C. The data presented here are indicative only, particularly for estimates of values for the extractive activities, i.e., community-based non-timber forest products (NTFPs) and logging, and there is clearly a need for additional multidisciplinary studies of the direct opportunity costs to smallholders of foregoing forest conversion.

In cases (largely outside Sumatra) where natural forest remains, how much is the timber worth? The ASB results so far suggest the greater payoff to additional information would be improved estimates for timber values, since they are many times larger than the NTFP estimates. These would involve techniques already developed (by ASB and others) to conduct the necessary biophysical and economic analyses. This is straightforward, at least from a static perspective. An equally important, but more difficult, question concerns option values for future development. How, for example, to factor in the option value of technological innovations, such as rubber clones, that are available, but not yet widely adopted?

Although community-based extraction of NTFPs may offer attractive returns to labor far exceeding rural wages (Tomich et al., 1998a), the relatively low returns to land (Table 2), well below those of rubber agroforests, suggest that this is not a feasible alternative for large numbers of people, because there is not enough land for everyone to practice this extensive livelihood strategy. So, in addition to the question of technological change mentioned above, what compensation would be available for the loss of livelihoods and depression of wages that would be transmitted directly through restrictions on access to resources and indirectly through labor markets?

Another difficult question concerns national policy, i.e., who owns the timber? Since colonial times, the policy has been that timber (and other natural resources) are the property of the state. But if local communities are not compensated for timber values, will incentives be sufficient to secure forest conservation? Moreover, in many situations in Indonesia, there are competing claims over these forest lands—between and among large-scale commercial enterprises (loggers and plantation companies) and local communities.

If C offsets are designed to include the full opportunity costs to these groups, there are additional thorny issues: which group (or groups) to compensate and how to apportion shares among competing claims? Paying two or more parties full compensation for timber and non-timber values plus foregone opportunities for conversion to other uses would undermine the cost-effectiveness of forest conservation projects in Indonesia as a means for C storage. Yet, as discussed below, failure to address conflicting interests can severely undermine the prospects for 'permanence' of these C stocks.

Smoke Signals (Im)Permanence of C Storage in Indonesian Landscapes

Differences in time scales of C storage between fossil C (eons) and C in land uses (decades) are a major sticking point on inclusion of terrestrial C sinks in the CDM, at least under the present Kyoto Protocol requirements that 'certifiable emission reductions' be real, additional, verifiable and permanent. Chomitz (2000) identifies this issue of permanence as the main feature distinguishing LULUCF alternatives from the mitigation measures in the energy and industrial sectors already included in the CDM. Forest and land fires dramatically illustrate the dubious permanence of terrestrial C sinks. In the case of boreal forests, fire suppression to increase C storage may even be counterproductive, since these measures ultimately can increase the risk and the intensity of fires through accumulation of fuel loads (Lashof and Hare, 1999, p. 107).

For Indonesia, Dauvergne (1998) estimated that the C emissions of the extraordinary fires in 1997 exceeded the annual emissions of western Europe's power stations and automobiles. The widely cited 1998 study by the Economy and Environment Program for Southeast Asia (EEPSEA) and World Wide Fund for Nature (WWF) estimated that of over $4.4 billion in damage from Indonesian fires and smoke in 1997, almost $3.8 billion (85%) was borne by Indonesia itself. Of this, the imputed value of the roughly one gigaton of C released was by far the biggest cost external to Indonesia (EEPSEA and WWF 1998). The EEPSEA/WWF estimates indicate that Indonesian citizens suffered the most short-term health effects by far ($924 million out of a total estimate of just over $940 million; about 98%). Although typically labeled a 'regional' or 'transboundary' problem, the most immediate environmental costs are felt by Indonesians. Unfortunately, because of political and social turmoil, it appears that the capacity of the government of Indonesia to act in the national interest on this issue is (at least temporarily) even weaker now than during the previous smoke 'crisis' in 1997/98 (Makarim, 1999). Thus, it would seem that

this smoke, which first and foremost is a threat to public health in Indonesia, results from lack of political accountability at local and national levels, not simply a lack of international environmental accountability.

What does this emission signal regarding permanence of C sinks in Indonesia? It is necessary to beware of simple diagnoses for the smoke that has plagued South-East Asia. This year, and every year, many causes underlie thousands of fires in Indonesia. Some of these are accidents, made worse by drought in some years (like 1997/98), but the majority of the fires in Sumatra in 1997 probably were not 'forest fires', nor do many of them appear to have been 'wild fires'. Analysis of satellite images of the 1997 fires in Sumatra show that few of the 'hotspots' in the satellite images—rough indicators of the occurrence of fires—were in natural forests in the peneplains, piedmont or mountain zones; most of the hotspots occurred in swamp forests and logged-over secondary vegetation (Stolle and Tomich, 1999). These areas with the most fires also are the prime areas for conversion. Thus, a large share of the 1997 fires appear to have had a purely practical motive, i.e., that fire is the cheapest way to clear land.

Fire is a tool of sustainable land management

Slashing and burning is the preferred method of land clearing in the tropics—for smallholders and large companies alike—because it is cheaper, at least from a private perspective, and easier than available alternatives. In addition, fire eliminates field debris, reduces problems with weeds and other pests and diseases, makes nutrients available in the form of ash and loosens the soil to make planting easier (Ketterings et al., 1999). In some ways it is preferable environmentally compared to some other land-clearing methods. For example, bulldozers and other heavy machinery cause soil compaction and erosion. Fire has many distinct advantages for low-income households: all other methods are more expensive; fire kills crop pests and diseases; and the ash provides much-needed fertilizer. Results of field experiments and surveys in Sumatra by Ketterings et al. (1999) demonstrate the key role that phosphorus (P) plays in resource management. Burning at low or medium intensity releases P into soil solution, but high-intensity fires reduce P levels. With greater fuel supply, hence hotter fires, the negative effects of heat on soil mineralogy offset the positive effects of additional ash.

Eliminating smoke does not necessarily cut carbon emissions

Land-clearing techniques exist that do not produce smoke, including biological methods to accelerate decomposition and various mechanical techniques that chip or shred biomass, either for mulching on the site or for transport off site for disposal or sale. All of these 'no-burn' techniques presently are more expensive than burning. Moreover, as noted above, burning produces many benefits for land users. Research may be able to reduce the economic and technical disadvantages (compared to burning) of certain techniques, such as mulching. If they are feasible, policies to regulate the timing of burning and new techniques for land clearing without burning could—over time—address part of the regional smoke problem. However, it must be emphasized that unless a significant portion of the biomass from land clearing is used to manufacture wood products with a long useful life (instead of simply decaying), these measures will only have a short-term effect on C emissions. Moreover, the attractiveness of technological alternatives to clear land without burning, or the level of subsidies required for adoption of these techniques, also are influenced by national policies, such as timber export taxes, discussed below.

Fire also is a weapon—this smoke signals policy and institutional problems

Fire is also the tool of choice for land clearing by large-scale agricultural and forestry companies in Indonesia—for many of the same reasons. Large-scale operators have also been known to burn land to drive out smallholders. Smallholders have been known to burn trees established by large-scale operators to retaliate for various injustices. Conflicts over land, resulting from unclear and insecure property rights, and land allocation policies that take too little account of established, albeit informal, local claims are at the heart of this problem. Aside from contributing to social conflict, 'land grabs' by large-scale operators that displace local people also undermine incentives at the community level to prevent, report and fight fires. If land-allocation policies concentrate holdings while destroying incentives for 'on-the-spot' fire prevention and management by the local people, this will increase the risk of future smoke 'crises'. And in this sense, the smoke problem can also be seen as a symptom of development strategies that favor large-scale enterprises, while ignoring legitimate local land claims. Addressing this widespread problem of land conflicts in Indonesia's Outer Islands, which contribute to the use of fire as a weapon, could be part of a longer-term strategy to reduce fire risks and to reduce poverty by spreading the benefits of regional development.

Knowledge gaps regarding permanence of carbon offsets in Indonesia

Real community-level participation and adequate incentives for local people are necessary (but not sufficient) conditions for permanent, secure C storage in Indonesian landscapes. Fay et al. (1998) have documented a pilot effort in this direction, but much remains to be learned about how to establish and replicate local participation and accountability across a huge archipelago characterized by extreme variation in ecology, culture and socio-economic conditions.

Some of Indonesia's fires (and the resulting smoke) are simply a symptom of the high returns to conversion of natural forest described above. Analysis of the 1997/98 fire event in Sumatra suggests that even more smoke is due to the agronomic and economic advantages of fire as a tool in sustainable management of profitable land-use systems. Although none approach the C-richness of natural forests, a number of these tree-based systems that involve periodic clearing, burning and replanting, represent significant C stocks (Table 2).

Even if only 'temporary' C storage spanning 2–3 decades is possible at the plot level of management of these systems, increasing average C in the landscape decreases C in the atmosphere. Additional C storage results from shifts in management of these heterogeneous land uses. For example, shifting from periodic burning to gap replanting poses greater challenges to measure and verify changes in C in a cost-effective way than the case of forest conservation discussed above. Also, it is not clear that a 25–30 year cycle of C accumulation and release at the plot level would meet the permanence requirements of the Kyoto Protocol, even if modifications of such systems resulted in increased C stocks at the landscape level. As one approach to accommodating the Kyoto Protocol's permanence requirements to the realities of smallholder land uses, Chomitz (2000) and others have proposed a ton-year approach, i.e. paying for a flow of C storage services rather than purchasing C stocks in perpetuity, to adjust C credits to duration and also to reward longer deferral of emissions. Thus, if approved within the Kyoto Protocol and if practical feasibility could be established, this approach could be a way to address a number of areas of social, economic and technological uncertainty; in effect, land-use options could be reviewed and adjusted annually. Although the ton-year approach has been a topic of interest to researchers, it has received little attention from politicians and policymakers involved in international negotiations, and little is known regarding the feasibility of implementation.

Carbon Offsets for Development

A special report of the Intergovernmental Panel on Climate Change (Watson et al., 2000) identified land-use change from cropland and grassland to agroforestry as the largest among potential LULUCF sinks for C globally. There are at least two major barriers to smallholder agroforestation for production of timber and other 'forest' products in Indonesia. First is tenure insecurity for millions of smallholders, because of conflicting claims on land that no longer is natural forest. A long-term process will be needed to develop workable and enforceable agreements between government and local communities regarding land use and production-sharing rights and responsibilities on these lands. The second barrier involves disincentives to smallholder production created by trade and marketing restrictions that undermine incentives regardless of where production takes place, even on private land. The benefits of deregulation of trade and marketing could be felt immediately by millions of smallholders throughout the country, and also represents the easiest step available for immediate reduction in C emissions from land-use change.

Trade restrictions undermine incentives for agroforestation

Indonesian forestry policies have been designed to depress domestic prices of timber relative to world prices. Policies that depress prices of wood products increase the 'waste' that must be disposed of by burning or other means. If these policies were eased (or removed), more of the wood felled in land clearing would be sold for timber, thereby reducing the amount that is burned. Significant increases in exports of agroforestry timber—from wood that currently is wasted—would help Indonesia earn foreign exchange and would reduce C emissions. Elimination of disincentives to smallholder production by deregulating agroforestry tree species is an important and administratively easy first step toward realizing farmers' potential contributions to meeting growing commercial demand for forest products and to agroforestation of deforested watersheds.

Some of the most complex forestry policy questions concern management of Indonesia's 'old growth' natural forests discussed above. But there are a significant number of agroforestry tree species grown by smallholders (and by large-scale estate plantations) that are not natural forest species. Complete deregulation of these agroforestry species thus poses no threat to Indonesia's natural forests, simply because these species are not found in Indonesia's old-growth natural forests. Three categories of agroforestry species can be distinguished to clarify the ecological and economic roles of each species for smallholders:

- **Exotic species.** None of these species are found in Indonesia's natural forests. Consider rubber wood, which is a substitute for ramin (*Gonystylus* spp.), one of the most valuable natural forest species. With the depletion of ramin, rubber wood has emerged as an important by-product of natural rubber production. Teak, mahogany and all but one of Indonesia's pines (*Pinus merckusii*) also are exotics. Although presently grown mainly in large-scale plantations dating from the colonial era, smallholders are interested in planting these species (often beside roads and along fence rows), despite their relatively long gestation periods, because of the high value of their timber.
- **Indigenous multipurpose species.** Coconut (*Cocos nucifera*) is the most widespread of these common species. Farmers mainly plant them for non-timber products, but timber is a valuable by-product at the end of the tree's productive life. These species are grown in large quantities by smallholders. Although these are indigenous species, most trees of these species now are planted and only a small proportion of these trees are found in Indonesia's natural forests.
- **Indigenous 'pioneer' timber species.** Although indigenous to Indonesia's natural forests, these fast-growing, light-loving species specialize in gap filling and, hence, are rare in old-growth natural forests. Their ecological niche also means they are well suited to domestication and planting in farmers' fields.

Current regulations covering trade and marketing of timber and other 'forest' products are designed for natural forest products, but are inappropriately applied to agroforestry products, which are produced from farmers' own labor, land and capital. These policies, which penalize smallholders who grow trees on their farms, include export taxes, natural resource rents and royalties, and administrative procedures for harvesting and transporting timber and other products that have been misclassified as 'forest' products. In addition to being an

administrative burden, the current complex felling and trading procedures for timber and other products grown on farms create various opportunities for illegal levies. The result is increased marketing costs, which reduce or eliminate farmers' profits. Particularly because these products all require substantial time and investment to produce, inappropriate application of these regulations makes production of agroforestry products, including timber, much less attractive than farmers' other alternatives.

Unlike trees in natural forests, agroforestry species are planted and managed by smallholders just like agricultural commodities. Removing current regulations on harvesting and trade of timber for agroforestry species would significantly improve incentives for development of Indonesia's smallholder farm forestry subsector. This would be an important step toward realizing the potential of smallholders to make a bigger contribution to meeting growing commercial demand for timber. Deregulation of agroforestry species would raise the economic benefits of growing trees on degraded lands and provide a new stimulus for farmers to improve productivity of lands that have been marginal for agricultural production. Therefore, in addition to reducing timber waste (and resulting C emissions), deregulating harvesting and trade in agroforestry species would help promote agroforestation and thereby produce environmental benefits on a local, regional, national and global scale.

Potential for agroforestation of grasslands

We now turn to a specific opportunity for agroforestation: Indonesia's extensive *alang-alang* (*Imperata cylindrica)* grasslands. More than 8.5 million ha (about 4% of Indonesia's land surface) are covered by sheet *alang-alang* (Garrity et al., 1997). Evidence from Indonesia and elsewhere in South-East Asia demonstrates that local people invest on their own to plant trees for timber and other products if they have secure claims over the products; natural risks, such as fire, are not too high; and they have access to markets.

Because changes in land use from grasslands to tree-based systems bring big changes in C stocks, compensation for C sequestration through investment to establish trees can significantly raise profitability of grassland conversion. Although there is no substitute for project appraisal for specific settings, significant imputed values of C sequestration from conversion to *Acacia mangium* and rubber agroforestry (under technologies, prices and policies prevailing in Indonesia) hold up even for a very conservative estimate of the value of C (US$5–20 per Mg C).

In fact, while compensation for C sequestration would raise the profitability of these investments significantly, this compensation would simply raise profits on investments that could have been undertaken anyway (Tomich et al., 1997). The calculations in Tomich et al. (1997) were based on preliminary data on above-ground C stocks, straight-line accumulation and discounting of annual C increments, and very conservative estimates for the value of C sequestration services (US$5–20 per Mg C). Better data on time-averaged C (Table 2) and higher values for sequestration services (say US$25 per Mg) strengthen these conclusions. (Recall that in the profitability estimates in Table 2, foodcrop systems on *Imperata* grasslands were unprofitable.)

These analyses suggest that, regardless of the future for the CDM, Indonesian initiatives aimed at *Imperata* grasslands should focus on establishing clear, secure tenure over products of alternative land-use systems and on removal of institutional failures and policy distortions. Under present circumstances of insecure tenure, local people lack incentives to control fires or to plant trees. A policy to establish secure property rights over all products—including the timber—for smallholders who convert plots of grassland by planting and managing trees could be an important first step in addressing the lack of tenure security and in creating incentives for community-based fire control. Without this local cooperation to control fire, sustainable rehabilitation of *Imperata* grassland is extremely difficult. Profitability of grassland conversion can also be destroyed by a number of policy distortions, including trade and pricing policies that depress output prices. Eliminating current disincentives to *Imperata* grassland conversion, such as the timber marketing restrictions discussed above, should come before any policy intervention to promote conversion. Property rights over all products, including timber, would create incentives necessary for local people to do the hard work to re-establish trees on grasslands. If tree planting is profitable, local people will do it once they are convinced they will reap the rewards of their work. If it is not profitable, the land will stay as it is.

Concluding Questions

'Carbon trading', payments for C sequestration services or for mitigation of C emissions, under the multilateral CDM or similar bilateral arrangements may develop as new mechanisms to compensate local communities in order to shift incentives toward more C storage in tropical landscapes. But many gaps remain, ranging from general questions about the meanings of key terms, for example, whether or not forest conservation and management of soil C could be included in C credits for 'deforestation', 'reforestation' and 'afforestation' (Sedjo et al., 1998), to a host of specific operational details of activities for which few countries have had relevant experience, apart from Costa Rica's path-breaking efforts (Chomitz et al., 1999).

ASB researchers are making progress in refining measurements of C stocks in different land uses at a policy-relevant scale (van Noordwijk et al., 2000), which may be adaptable to establish and verify real changes in C stocks within a project or program for C offsets. As reported here, ASB has also produced estimates of the 'farmgate' or 'forestgate' direct payments for C sequestration that would be necessary to shift smallholders' incentives from privately profitable but less C-rich systems to land-use systems that store more C; however, these should be treated as indicative of relative magnitudes rather than definitive results that could be used as a basis for design of interventions. For both forest conservation and agroforestation, studies are needed across a wider range of circumstances found in Indonesia, to improve estimates of the direct opportunity costs of shifting to, or conserving, land uses that can store more C, and to verify the C increments associated with those changes.

A key policy question for Indonesians

Even though available estimates are rough, it is unlikely that further studies will reverse the result reported in this paper that timber values far exceed values of non-timber forest products in Indonesia. Better estimates of opportunity costs are needed for the value of timber in the remaining 'pristine' natural forests and, perhaps more importantly, in community-managed forests. The commercial value of this timber appears to be a decisive element in the cost-effectiveness of C offsets for forest conservation in Indonesia, compared to other regions (especially the Amazon). The question of timber values also has a deep political dimension in Indonesia, i.e., who owns the timber: the central government, provincial governments, local communities or individuals? This has important implications for implementation of C offsets for forest conservation. It is up to Indonesians how (and when) this policy question will be answered. In the meantime, C offsets through agroforestation seem more politically feasible than C offsets through forest conservation in Indonesia, because property rights over timber from planted trees are easier to establish and resolve than property rights over timber in natural forests.

Institutional questions for designers of C offsets

Despite the present limits of our information on changes in C stocks and opportunity costs of alternatives, even less is known about the actual 'cost of doing business' (the transaction costs) when smallholder communities (and their heterogeneous landscapes) are involved in C trade. These costs are important: if they are too high compared to the global price of C stocks, smallholders' incentives will be inadequate to induce a change in behavior. At present, the transaction costs of trading C between OECD countries and smallholder communities in the humid tropics may well be higher than transaction costs of C trading within OECD countries. Is there scope to reduce the transaction costs of trading C with smallholder communities through 'learning by doing'? Some specific action research questions include:

- What are the relative costs and effectiveness of alternative institutional arrangements (payments for C in perpetuity versus pay-as-you-go ('ton-years'); direct versus indirect incentives)? It would be desirable to disaggregate components of transaction costs in these studies, i.e., search costs, negotiation and contracting costs, intermediation and marketing costs, monitoring and enforcement costs, and costs of specific interventions to target poor households, to identify the largest elements of costs of trading with smallholders and the best opportunities for cost reduction.
- Is it feasible to develop and use regional baselines to measure and validate C stocks for specific smallholder landscapes to reduce costs of validation and monitoring?
- Can methods be developed for projecting dynamic baselines—with and without project scenarios that reflect economic and environmental shocks, structural transformation and (possibly) technological change? Can household, regional or quasi-general equilibrium approaches—extensions of techniques developed by members of the ASB Consortium—be validated and transferred for use by project managers, policymakers and negotiators? Will these simulation approaches be acceptable to C trading partners?

On these issues, there is much to be gained from an international synthesis to identify methods for accelerating the learning process and thereby bring down the 'transaction costs' of dealing with smallholder communities. There is already experience from the 'activities implemented jointly' (AIJ) projects of the pilot phase of the UNFCCC that could produce relevant lessons (MacDicken, 1999). In their review of AIJ experience in Brazil, India, Mexico and South Africa, Sathaye et al. (1999) conclude that there are major issues to address regarding the institutional basis for establishing additionality of C (baselines and leakages) and also the additionality of funding (fungibility of ODA versus new money for C offsets). Whether it makes sense for Indonesia to participate in this process of institutional innovation also depends on these questions regarding additionality of C and funding.

Is the carbon 'additional'?

It may be too soon to discern whether (or not) the CDM and similar C offset schemes hold attractive opportunities to combine global environmental benefits with development in Indonesia. However, some insights may be gleaned by considering whether examples discussed in this paper might be eligible for C offset credits regarding three key criteria:

1. the increase in C stocks must be 'real' (measurable changes) and 'additional' (not an increase that would have happened anyway under 'business as usual');
2. there must be a quantifiable 'baseline' for comparison (some measure of what would have happened without payment of the C credits); and
3. 'leakages' (offsetting effects that reduce the net change) and 'unintended consequences' (perverse incentives that produce counterproductive results) must both be considered in appraisal of potential projects.

Note also the presumption that projects, bounded in time and space, are the mechanism for implementation. To our knowledge, there has been no discussion of C credits for broader programmatic approaches (investments in infrastructure or scientific research and development) or of policy reforms. Each of these broader interventions could rival the project approach in potential to shift incentives in favor of C sequestration.

Consider the timber trade policy reforms discussed above. Eliminating these restrictions would have real, measurable effects, with trade statistics providing a basis for developing a baseline. However, since it appears (at least to us) that it clearly is in Indonesia's national interest to reform these policies anyway, additionality of this C may be questionable. So, ironically, the measure that would have the most immediate effect on Indonesia's C emissions may not even qualify for C credits.

This points to a more general paradox: no pure 'win–win' opportunity would appear to be eligible, since presumably it would be done anyway. Thus, although Sedjo et al. (1998)

singled out 'afforestation on low cost lands as among the less expensive alternatives when compared with other greenhouse gas reduction policies', the analysis of smallholder agroforestation of *Imperata* grasslands summarized above indicates that these investments could be profitable (even without C credits) if necessary policy and institutional reforms were feasible. (Once such reforms were implemented and all the opportunities that were profitable without C credits were exhausted, then presumably it would be possible to justify C credits to promote further grassland conversion.)

In this respect, C offset projects are no different from any other development project in that effects of policy distortions and institutional weaknesses need to be considered in project appraisal. Ideally, appropriate institutional and policy reforms would precede C offset arrangements. Indeed, Indonesia could choose to hasten this process in order to prepare for C offset opportunities. But how likely is that to happen when there are so many other pressing social, economic and political challenges facing Indonesia's policymakers? Although removing existing constraints and disincentives that hamper agroforestation by smallholders would benefit Indonesia, would it not also be justifiable—in terms of real, additional, verifiable C storage through sustainable development—to fund assistance to break these policy and institutional bottlenecks through C payments for enabling policy and institutional reforms? Of course, this is precisely the domain of conventional international development assistance by agencies such as the World Bank, regional development banks and the overseas development assistance provided by individual OECD countries, which brings us to our final question.

Will the funds for carbon offsets be 'additional'?

An international agreement to establish and enforce quotas for the rich countries responsible for most C emissions would be the driving force behind private-sector participation in C offset trading, and the source of billions of dollars of additional funding for development. Without binding quotas on C emissions, the most tangible rationale for additional private finance would fail to appear. In a world without clear prospects for an international agreement like the Kyoto Protocol, would private companies choose to devote significant funds to C offsets? While there may be some scope for companies to generate tangible returns on these investments through favorable public relations, the concern is that 'funding could be seen as simply fulfilling previously broken promises of aid rather than supplying truly additional support' (Sathaye et al., 1999, p. 193). In a study of AIJ experience by La Rovere (cited in Sathaye et al., 1999, p. 193), the author 'offers a rather dismal description of the whittling down of several hundred million dollars of aid promised by G–7 countries through the Global Environmental Facility (GEF) and other donors. The promised sum was eventually reduced to a fraction of the original commitment'. Agroforestation could be vulnerable target for re-labeling of conventional development assistance as 'clean development'. At best, the results of this re-labeling would be neutral. But, given what promise to be extraordinarily demanding institutional requirements for implementation of CDM-type arrangements, there is a real risk of crowding out initiatives in other sectors (public health, education) that have higher priority in national development strategies. Thus, the bottom line is that significant commitment of additional private funds appears to be necessary if C offsets are to provide real benefits for Indonesia and other tropical countries.

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Carbon stocks in Indonesian homegarden systems: Can smallholder systems be targeted for increased carbon storage?

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Abstract. *Homegardens are a common smallholder agroforestry system in Indonesia and throughout the tropics. These species-rich, tree-based systems produce non-wood and wood products for both home use and market sale. Due to their high biomass, these systems simultaneously offer potential for carbon (C) storage. While small size limits the amount of C stored by individual smallholder agroforestry systems, on a per area basis these systems can store as much C as some secondary forests. In aggregate, smallholder homegarden agroforestry systems can contribute significantly to a region's carbon budget while simultaneously enhancing smallholder livelihoods. A field study in Lampung, Indonesia indicates that homegardens with an average age of 13 years store 35.3 Mg C ha^{-1} in their above-ground biomass, which is on par with the C stocks reported for similar-aged secondary forests in the same area. However, to compare accurately the C stocks of different land-use systems a scale is required that adjusts C stocks of the systems' ages and rotation lengths to a common base. The time-averaged C stock, which is half the C stock at the maximum rotation length, serves this purpose. Our projections reveal that, depending on management options, the time-averaged above-ground C stocks of homegarden systems could vary from 30 to 123 Mg C ha^{-1}. These projected time-averaged above-ground C stocks of homegardens are substantially higher than those of* Imperata*-cassava systems (2.2 Mg C ha^{-1}), which is an extensive vegetation type in the study area. If homegarden systems and other smallholder tree-based systems were to expand in currently degraded and underutilized lands, such as* Imperata *grasslands, the C sequestration potential would be about 80 Mg C ha^{-1}, with considerable variation depending on species composition and management practices. Clear opportunity exists to induce management that leads to higher C stocks at the systems level. However, incentive mechanisms are needed that assure smallholders will benefit from selecting management practices that favor higher C stocks.*

Key words: agroforestry, agroforestation, carbon sequestration, certified emissions reductions (CERs), clean development mechanism (CDM), climate change, greenhouse gas, *Imperata* grasslands, land rehabilitation, tree farming

Introduction

Increasing levels of atmospheric 'greenhouse gases' are generally accepted to be a main contribution to global warming, which, studies indicate, is changing the Earth's weather patterns and could raise ocean levels substantially in the next 100 years (Schimel et al., 1995; Watson et al., 1996). These climatic changes can impact environmental norms and human populations, causing serious negative disturbance to the global economy. As international agreements over greenhouse gas emissions and global warming are negotiated, there is growing interest in the possibility of reducing the increase in the amount of carbon dioxide (CO_2) in the atmosphere through forest-based carbon (C) sequestration projects.

Forest-based land-use systems—natural forests, forest plantations, and agroforestry systems—sequester CO_2, through the C stored in their biomass. By promoting land-use systems which have a higher C content than the existing plant community, net gains in C stocks (hence sequestration) can be realized. The most significant increases in C storage can be achieved by moving from lower-biomass land-use systems (e.g., grasslands, agricultural fallows and permanent shrublands) to tree-based systems. Forest-based C storage projects have been implemented by electric utilities and other industries as a means to offset the C released by their use of fossil fuels. These projects are still in their preliminary phases. As many efforts to achieve increased forest C storage may have negative implications for the rural poor, options that support human livelihoods deserve special attention. Projects implemented under the Clean Development Mechanism (CDM) of the Kyoto Protocol will offer opportunities for investors seeking 'certified emissions reductions' (CERs) to invest in developing countries for the dual objective of reducing greenhouse gas emissions and contributing to sustainable development. Forest-based C storage projects will likely be part of the CDM.

Indonesia provides an attractive environment for C investment. There are over 8.5 million ha of *Imperata* grasslands in Indonesia (Garrity et al., 1997). Originally forests, these lands

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include pure grasslands, cyclic fallows and shrublands, and are acknowledged to be underutilized. There is clear interest, at both governmental and smallholder farmer levels, to convert some of these lands to a more productive land-use, including tree-based systems (Tomich et al., 1997). Homegardens are a common smallholder agroforestry system in many parts of Indonesia. These species-rich tree-based systems usually occupy lands immediately surrounding the household and are used to produce a diverse array of food and other products. Traditionally intended to produce goods mainly for home consumption, the advent of rural infrastructure and market-economies has made homegardens more commercially oriented. Homegardens production now commonly serves both household and market demand, providing families with much needed income (Krol, 1992; Michon and Mary, 1994).

Simultaneously, homegardens, and other tree-rich smallholder systems, offer potential for C storage because of their high, woody biomass. The question we raise here is: can the role of smallholder agroforestry systems be expanded to serve global environmental goals by targeting them for increased C storage? The objective of this study was to generate C stock inventory data for homegarden systems in Lampung province, Sumatra, Indonesia. The study results are compared to C stock data for other land-use systems in Sumatra previously obtained by the global Alternatives to Slash-and-Burn (ASB) project (Tomich et al., 1998).

Methods

Study area

The study was conducted in the villages of Karang Sakti, Negara Jaya and Tegal Mukti in Pakuan Ratu district in Lampung province, Sumatra, Indonesia at one of the benchmark areas of the ASB project. Soils are well drained, deep (>1–1.5 m), acidic and of low fertility. Aluminum toxicity is common, especially below a depth of 15 cm. Root development is possible down to 1–1.5 m. The major soil groups are Oxisols/Ultisols and Inceptisols covering 64 and 29% of the area, respectively. Elevation is less than 100 m above sea level, mean annual temperature is 28°C, varying between 22 and 33°C (van Noordwijk and Purnomosidhi, 1995; van Noordwijk et al., 1996). Annual rainfall averages 2200–2500 mm, with 5–6 months greater than 200 mm and 1–4 months less than 50 mm.

The study site is a government-sponsored transmigration area for the relocation of spontaneous migrants who previously settled in mountainous areas of Lampung. Transmigrant households were each given 2 ha, including a home lot of 0.25 ha where homegardens were typically developed. The species composition of local homegardens include trees that produce fruit, vegetables, spice, oil, medicine, other non-wood products and timber; perennial understory plants that produce non-wood products; and annual crops—such as vegetables, cassava (*Manihot utilissima*), maize (*Zea mays*) and rice (*Oryza sativa*) (Gintings et al., 1996). The other major land-use classes in the area are sugarcane (*Saccharum officinarum*) plantations, cassava (grown as a cash crop for local agribusiness), other agriculture crops, *Imperata* grasslands and degraded secondary forests.

Site selection

Detailed land-use maps at the smallholder level were not available for the study area. Homegardens were selected if the farmer gave permission and the structure and species were considered typical of local homegardens. Homegardens were excluded if they contained 50% or more of: (1) annual crops (vegetables, cassava, maize, rice, etc.); or (2) one market-oriented tree crop (e.g., coffee (*Coffea robusta*), coconut (*Cocus nucifera*), sengon (*Paraserianthes falcataria*), etc.). Homegardens that contained large areas (25% or more) of rice paddies or fishponds were also excluded. Approximately 25–30% of the assessed homegardens were rejected for these reasons.

Carbon monitoring system

The carbon monitoring system used in this study was developed by Winrock International's Carbon Monitoring Program to quantify the amount of C in land-use systems using forest and agroforestry inventory principles and practices (MacDicken, 1997). Winrock's system quantifies C sequestered by measuring changes in four main carbon pools over time or comparing the C in these four pools with other land-use options. The main C pools are: above-ground biomass, litter, herbaceous material and soil. The Winrock methods are very similar to those used in the ASB project (Hairiah et al., 1999; Palm et al., 1994).

Plot installation and measurement

At each farm site, homegarden dimensions were measured and uncorrected global positioning system (GPS) coordinates taken. Most homegardens were rectangular in shape and roughly 75 m × 25 m. The center point of each homegarden was located and two subplots were laid out perpendicular to the longest borders, along a line that bisected the center point, half the distance between the center point and the short borders. Figure 1 provides a schematic of the sampling design used to locate the center point and subplots. In the few cases when homegardens were L-shaped, a third subplot was established in the middle of the short leg of the L using a modification of the center point location process described above. The agroforestry inventory methods developed by MacDicken (1997) recommend a minimum of four subplots be installed per site, but the homegardens in this study were too small to accommodate four subplots. Two subplots per homegarden provided a sampling intensity of approximately 14%.

Each subplot was a circle with a radius of 8.9 m (area 249 m^2). The diameters of all trees in each subplot with a diameter at breast height (DBH 1.3 m above the soil surface) greater than 5 cm were measured and the species recorded. For coconut and banana (*Musa* spp.), species not covered by the standard diameter–biomass relationship, in addition to DBHs, heights were measured using a clinometer. DBHs of down but intact trees—either living or dead—were also recorded. From the subplot center, four points were established (north, east, south

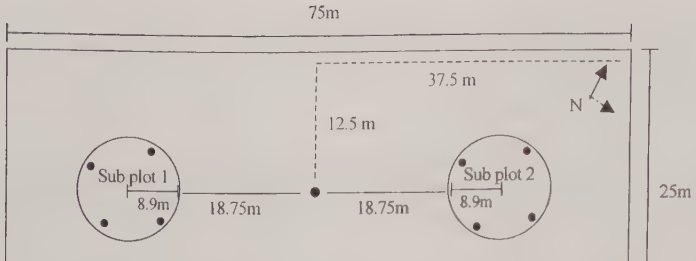


Figure 1. Schematic of sampling design used in homegarden plots. The center point and subplot are located by the following process. From the north-east corner, travel half the distance of the average length of the northern and southern borders (37.5 m in this example). From that mid-point of the northern border, travel half the distance of the average width of the eastern and western borders (12.5 m) along the bearing of the eastern border to the plot center point. Install two subplots along the line that bisects the center point, by traveling half the distance from the center point to the eastern and western border (18.75 m). Subplots are circles with radius 8.9 m. Four sample points are established on north, south and west bearings, 1 m inside the subplot boundary.

and west) 1 m inside the subplot boundary. At these four points samples were collected of herbaceous vegetation (all living plants with a DBH of less than 5 cm diameter), litter (all organic matter above the soil surface having a diameter of less than 5 cm) and soil (to 30 cm depth).

The herbaceous and litter samples were collected by placing a circular aluminum sample ring (0.28 m^2) at the sample points. All living material within the ring was cut at the soil surface. To minimize damage to farmers' crops, neither juvenile trees nor agricultural plants within the sampling ring were destructively sampled for inclusion in the herbaceous sample. This missing biomass was not estimated because: (1) the situation occurred infrequently; and (2) the herbaceous pool contains only a fraction of the total C stock of a forest-based land-use system. All herb and litter samples were weighed using a spring scale. Herb samples were mixed and subsampled for moisture content determination. The same process was used for litter subsampling. Soils samples were collected within the aluminum ring after all herb and litter materials were collected by digging a 30-cm pit and slicing a sample from the pit wall from 0 to 30 cm depth. Sample size was approximately 1 cm thick, 10 cm wide and 30 cm long. Soils were sieved through a 5-mm mesh screen, mixed to a uniform color and consistency and a subsample taken for C analysis. Walkley–Black analysis for soil organic C (Walkley and Black, 1934) was conducted at Brawijaya University, Malang, East Java. At one of the two subplots a soil bulk density sample was taken at a depth of 15 cm by hammering an aluminum cylinder into the pit wall. Bulk density was determined by drying the sample in an oven at 100°C for 24 hours. Soil C ha^{-1} was determined according the following formula: Walkley–Black value (percent C) × bulk density ($g\,cm^{-3}$) × 3000 $kg\,m^{-2}$ (MacDicken, 1997).

Estimating above-ground biomass

To estimate biomass (kg) of above-ground vegetation, the following general biomass regression equation was used for most species: $\exp\{-2.134 + 2.530 \times \ln(DBH)\}$ (Brown, 1997). The biomass of coconuts was estimated by calculating volume then converting to biomass by multiplying by an average density of 0.5 $g\,m^{-3}$. Biomass of banana was estimated by applying the general biomass equation and subtracting 50%. Root biomass was estimated by taking 25% of above-ground biomass (Cairns et al., 1997). Tree biomass is converted into C by multiplying by 0.5 (MacDicken, 1997).

Results

Carbon stocks

Measurements were made on a total of 19 homegardens. Homegarden ages reported by the landowners varied from 12 to 17 years, with an average age of 13 years. Tree age within the homegardens varied greatly as landowners develop homegardens over a number of years. Total C per homegarden ranged from 56 to 174 Mg C ha^{-1} with an average of 107 Mg C ha^{-1} (Table 1). Tree biomass (above-ground plus roots) and soil accounted for 98% of these C stocks (41% and 57% respectively). Above-ground C in the homegardens varied from 6.3 to 84.0 Mg ha^{-1}, with an average of 35.3 Mg ha^{-1} with a coefficient of variation (CV) of 60%. Soil C varied from 10.4 to 103.7 Mg ha^{-1}, with an average of 60.8 Mg ha^{-1} (CV of 32%). The remaining 2.2% of the C stock was in the litter (1.9%) and herbaceous (0.3%) pools.

Tree component

The homegardens were diverse, containing 45 tree species. A total of 597 trees were sampled, with an average of 34 per homegarden (2–3 plots/homegarden). The species, their predominance in the homegardens, and their primary uses are given in Table 3. Eighty percent of the species in the homegardens provide primarily non-wood products or services (NWPS)—fruits, vegetables, spice, oils, medicines, resins and soil improvement. Coincidentally, these species also account for 80% of the trees surveyed and 73% of the tree biomass (Tables 2 and 3). Twenty percent of the species in the homegardens, representing 20% of the trees sampled and 27% of the tree biomass, are grown primarily for timber and wood production (Tables 2 and 3). These species may also produce non-wood products or services, but these products and services are of secondary importance.

Inventory method

A shortcoming of the method used in this study is that subplot establishment is relegated to the interior of the plot. Thus, species that occur most often, or exclusively, near the borders are under-represented in the inventory. For example, bamboo was observed along the border in approximately one-third of the homegardens surveyed, but not in any of the subplots. A line-transect or rectangular plot may be preferable for small-sized smallholder agroforestry systems.

Table 1. Carbon stocks by main pools for 19 homegarden systems in Indonesia.

Plot	Age	Above ground	Litter	Herbaceous	Soil	Roots	Total
	(years)			Mg C ha^{-1}			
1	15	61.9	2.4	0.4	65.7	15.5	145.9
2	15	6.3	0.3	0.2	55.3	1.6	63.7
3	15	34.1	2.7	0.1	10.4	8.5	55.8
4	15	17.6	3.2	0.2	52.1	4.4	77.5
5	12	14.4	2.5	0.1	44.9	3.6	65.5
6	12	21.6	1.9	0.3	69.3	5.4	98.5
7	12	23.0	0.2	0.2	41.2	5.8	70.4
8	13	34.8	2.3	0.1	72.5	8.7	118.4
9	12	24.9	1.6	0.3	40.3	6.2	73.3
10	12	22.6	1.2	0.5	72.5	5.6	102.4
11	14	65.3	1.7	0.3	55.3	16.3	138.9
12	17	17.1	2.7	0.4	65.7	4.3	90.2
13	13	21.6	2.8	0.5	51.6	5.4	81.9
14	13	45.4	3.6	0.2	77.5	11.4	138.1
15	13	84.0	0.0	0.0	69.3	21.0	174.3
16	12	56.2	1.4	0.8	76.1	14.0	148.5
17	13	46.3	0.8	0.3	103.7	11.6	162.7
18	13	53.8	4.0	0.1	69.3	13.4	140.6
19	13	19.3	2.9	1.4	62.1	4.8	90.5
Mean (SD)	13.4	35.3 (21.0)	2.0 (1.2)	0.3 (0.07)	60.8 (4.4)	8.8 (5.3)	107.2 (37.2)
CV	-	60%	57%	95%	32%	60%	34.7%
% of total	-	32.9 %	1.9%	0.3%	56.7%	8.2%	—

Table 2. Number of trees, average DBH, non-wood species C, and wood species C for each of the 19 homegarden systems in the study.

Plot	Trees/ subplots	Average DBH[1]	Trees ha^{-1}	NWPS[2] (trees ha^{-1})	NWPS trees (Mg C ha^{-1})	Timber (trees ha^{-1})	Timber trees (Mg C ha^{-1})
1	42	15.7	840	460	56.4	380	21.0
2	13	17.7	260	260	7.9	0	0.0
3	25	15.0	500	380	27.0	120	15.6
4	33	10.6	660	520	15.0	140	7.0
5	40	10.1	800	580	9.0	220	9.0
6	24	13.8	480	460	27.3	20	0.7
7	59	10.8	1180	720	16.5	460	12.3
8	37	12.5	740	620	19.7	120	23.8
9	34	15.8	680	680	31.1	0	0.0
10	15	18.3	300	300	28.2	0	0.0
11	35	17.0	700	300	22.9	400	58.7
12	27	15.1	540	520	18.4	20	3.0
13	27	15.4	540	540	27.0	0	0.0
14	36	14.7	720	700	55.2	20	1.6
15	48	16.9	960	820	77.9	140	27.1
16	19	19.0	380	380	70.2	0	0.0
17	17	17.0	340	280	17.0	60	40.9
18	46	12.8	920	820	60.5	100	6.7
19	20	17.3	400	320	13.7	80	10.4
Average	31.4		628.4	508.4	31.6	720	12.5
% of tree carbon					72%		28%

[1] DBH, diameter at breast height.
[2] NWPS, non-wood products and services.

Table 3. Tree species identified in the 19 home garden systems.

Botanical (local) name	Primary uses[2]	Trees		Occurrence	
		Total no.	% of total	no. HGS[3]	%HGS
NWPS[1]					
Cocos nucifera (coconut or kelapa)[4]	ol, fr, vg, md, su, wd	66	11.1	17	89.5
Mangifera indica (mangga)[4]	fr, vg, md, wd, tn	59	9.9	15	78.9
Musa spp. (banana or pisang)	fr	57	9.5	9	47.4
Nephelium lappaceum (rambutan)[4]	fr, ol, st, md, wd, tn	55	9.2	9	47.4
Parkia speciosa (petai)[4]	sp, vg, md, wd	39	6.5	14	73.7
Archidendron pauciflorum (jengkol)[4] (syn. *Pithocellobium jiringa*)	vg, md, wd, tn	28	4.7	11	57.9
Artocarpus heterophyllus (nangka)[4]	fr, vg, md, dy, tn	26	4.4	12	63.2
Ceiba pentandra (kapok)[4]	co, dy, ol, fr, vg, md	18	3.0	7	36.8
Gliricidia sepium (gamal)	si, or, wd	13	2.2	3	15.8
Coffea robusta (kopi)	st	12	2.0	4	21.1
Erythrina spp. (dadap)	si, md, wd, or	12	2.0	3	15.8
Gnetum gnemon (melinjo)[4]	fr, vg, wd, dy	11	1.8	5	26.3
Hevea brasiliensis (karet)	rs, ol, fr, vg	10	1.7	3	15.8
Spondias spp. (kedongdong)	fr, vg, wd	10	1.7	3	15.8
Theobroma cacao (coklat)	ol, st, md	8	1.3	3	15.8
Syzygium aqueum (jambu air)	fr, md, wd	7	1.2	4	21.1
Anacardium occidentale (jambu mete)[4]	fr, ol, vg, sp, md, tn	5	0.8	3	15.8
Averrhoa bilimbi (belimbing)	fr, sp, md, su, wd	5	0.8	1	5.3
Garcinia parvifolia (kardis)[4]	fr, rs, wd	5	0.8	1	5.3
Leucaena leucocephala (lamtoro)	vg, si, st, wd, tn, or	5	0.8	2	10.5
Aleurites moluccana (kemiri)[4]	sp, ol, fr, md, wd, dy	4	0.7	2	10.5
Psidium guajava (jambu biji)	fr, ol, sp, st, md, su	4	0.7	4	21.1
Annona muricata (sirsak)[4]	fr, st, md, dy	3	0.5	2	10.5
Melia azedarach (mindi)[4]	md, ol, su, wd, rs, or	3	0.5	1	5.3
Artocarpus integer (cempedak)[4]	fr, vg, md, wd, dy	2	0.3	2	10.5
Persea americana (alpokat)[4]	fr, ol, wd	2	0.3	2	10.5
Annona reticulate (buah nona)	fr, md, wd, tn	1	0.2	1	5.3
Arenga pinnata (kolang kaling)[4]	su, fr, vg, st, md, wd	1	0.2	1	5.3
Cinnamomum parthenoxylon (kayulada)[4]	sp, md, wd	1	0.2	1	5.3
Flacourtia rukam (rukam)[4]	fr, vg, md, wd	1	0.2	1	5.3
Hibiscus tiliaceus (waru)	dy, md, wd, cf, or	1	0.2	1	5.3
Mangifera foetida (pakel)[4]	fr, vg, md, wd	1	0.2	1	5.3
Mangifera odorata (kurai)[4]	fr, md, wd	1	0.2	1	5.3
Tamarindus indica (kayu asam)[4]	sp, ol, fr, st, md, wd	1	0.2	1	5.3
Unknown (sanu)	–	1	0.2	1	5.3
	Subtotal	478	80.1		
Timber species					
Paraserianthes falcataria (sengon putih)	wd, si, dy	59	9.9	7	36.8
Acacia ariculiformis (akasia)	wd, si, or	18	3.0	3	15.8
Tectona grandis (jati)	wd, md, tn, cf	12	2.0	4	21.1
Acacia mangium (mangium)	wd, si, or	6	1.0	1	5.3
Alstonia spp. (pulai)	wd, st, md, rs	6	1.0	4	21.1
Peronema canescens (sungkai)	wd, med, or	6	1.0	1	5.3
Unknown (ladahan)	wd	5	0.8	2	10.5
Peltophorum dasyrachis (sengon merah)	wd, si, tn	3	0.5	2	10.5
Terminalia citrina (jaling)	wd, md, tn	3	0.5	1	5.3
Schima wallichii (puspa)	wd, md, tn	1	0.2	1	5.3
	Subtotal	119	19.9		
Total NWPS species		36	80.0		
Total timber species		9	20.0		
Total species		45	100		

[1] NWPS, non-wood products and services species.
[2] Key: co, cotton: cf, craftwood; dy, dye; fr, fruit; md, medicine; ol, oil; or, ornamental; rs, resin; si, soil improvement; sp, spice; st, stimulant; su, sugar; tn, tannin; vg, vegetables; wd, wood.
[3] no. HGS, number of homegarden systems.
[4] Long-lived NWPS species.
Sources: Mackey, 1996 (*Acacia mangium*), Pinyopusarerk, 1996 (*Acacia auriculiformis*), Leaving and de Foresta, 1991 (all others).

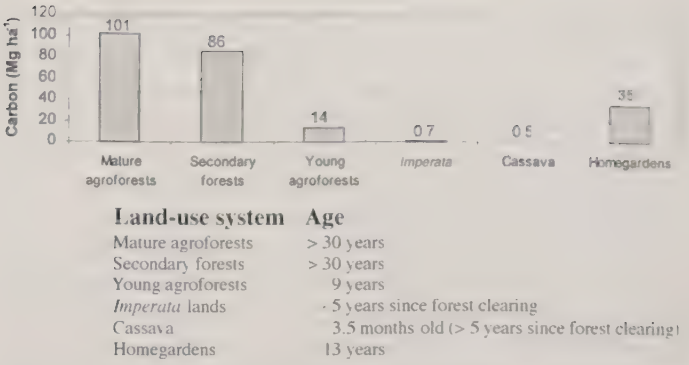
Discussion

There is great variation in the homegarden systems studied, particularly in the soil C, which varied from 10.4 to 103.7 Mg C ha^{-1}. This wide range of soil C results from various past and present management practices. Some of the homegardens studied were established immediately after the removal of secondary forests. Others were established after the land had been intensively cultivated for sugarcane and other seasonal crop production. Management of homegarden understories also varies greatly, from intense cultivation of annual crops (mainly cassava) to forest-like natural regeneration, pasture and even bare soil. Cleaning, weeding, burning and relocation of biomass are common management practices associated with intensive cultivation that lead to a steady loss in soil C. Conversion of natural forests or grasslands to permanent agriculture may result in 20–50% loss of soil C (Sampson and Scholes, 2000). However, the conversion of agricultural soils back to forest does not quickly reverse this loss (Detwiler, 1986). Soil C levels on such sites are expected to increase for decades or centuries (O'Connell and Sankaran, 1997, cited in Schlamadinger and Karjalainen, 2000). In recognition of this condition, the discussion that follows focuses on the tree biomass, as the pool where the greatest gain in C stocks can be achieved by reforesting *Imperata* grasslands or degraded agricultural lands.

Tree density varied from 13 to 59 trees sampled per homegarden (equivalent to 260–1180 Mg ha^{-1}). On average, above-ground biomass accounted for only 33% of the total C in the homegarden systems. Studies under the ASB project in the neighboring province of Jambi show that the portion of C in the tree biomass of tree-based land-use systems increases with age, up to 80% for a 120-year-old natural forest, which contains 500 Mg C ha^{-1} (Tomich et al., 1998). ASB studies in Lampung indicate that tree biomass accounts for 60–65% of total C in 30-year-old secondary forests and mature agroforests (Hairiah, 1997). Aggregated, data from the ASB project and this study show that all tree-based land-use systems accumulate similar C stocks over similar time periods. This indicates that, in terms of C sequestration, the homegardens, with an average age of 13 years, are still very young and will continue to accumulate C steadily for a long time, depending on management.

The C content of homegardens in this study compares favorably with the C content of five other land-use systems in Lampung (Hairiah, 1997). These five land-use systems are mature agroforests, secondary forests, young rubber agroforests, *Imperata* land and cassava fields. The inventory method used by Hairiah (1997) was similar to that used in this study. Both studies used the same general biomass regression equation. The main methodological difference occurs in measuring soil C, Hairiah (1997) used samples collected from a depth of 0–15 cm and our method uses samples collected from 0–30 cm. This is not a critical difference since soil C levels are peripheral to our primary analysis and discussion.

The C stocks for the six land-use systems in Lampung are highly variable (Fig. 2). In terms of above-ground biomass,



Land-use system	Age
Mature agroforests	> 30 years
Secondary forests	> 30 years
Young agroforests	9 years
Imperata lands	5 years since forest clearing
Cassava	3.5 months old (> 5 years since forest clearing)
Homegardens	13 years

Figure 2. Comparison of above-ground carbon stocks for various land-use systems in Lampung, Indonesia.

homegardens contain significantly more C ha^{-1} than *Imperata* grasslands, cassava fields and young rubber agroforests. These data indicate that homegardens had approximately 34.7 Mg C ha^{-1} more (58 times greater) than *Imperata*–cassava systems and 21 Mg C ha^{-1} more (1.5 times greater) than young rubber agroforests. *Imperata* and cassava systems maintain low woody biomass and C stock levels because of periodic fires or annual harvesting. Mature agroforests and secondary forest contain higher stocks of C than homegarden systems, by 66 Mg C and 51 Mg C ha^{-1}, respectively. This is due mainly to the presence of younger trees in homegardens.

One of the challenges of C stock estimate in homegarden systems (as well as other agroforestry systems) is accurate estimation of above-ground tree biomass. The above-ground biomass in this study was estimated using a general regression equation that was developed for tropical moist forests. The size of individual tree canopies in a forest could be smaller than those found in an open agroforestry setting, as the trees in some agroforestry systems have more space and access to light. The difference in structure could result in errors in our estimates. Likewise, trees found in homegardens could be misshapen if branches are cut for fuelwood or other uses. This could also lead to errors in our estimates. At the time of this study, local biomass equations were not available but will need to be developed in the future to reduce possible error in above-ground biomass estimation.

To accurately compare C stocks of different land-use systems a scale is required that adjusts the age and rotation length to a common base. Tomich et al. (1998) suggest using the time-averaged C stocks for comparison between systems. The time-averaged value is half a system's C stock at its maximum age (or rotation length). The time-average above-ground C stock of *Imperata*–cassava systems studied by the ASB project is about 2.2 Mg C ha^{-1} (Palm et al., 1999). For comparison, the time-averaged above-ground C stock of a homegarden at its maximum rotation length can be estimated by grouping the component species into rotation age classes, projecting linear growth to estimate tree C stock at the maximum rotation length for each class, and then taking half the above-ground C stock level for the homegarden at maximum rotation length. Based on the inventory data we select three rotation classes: long-lived NWPS species, medium-lived NWPS species and timber

species. Long-lived NWPS in Table 3 represent 57% of the homegarden species. Experience indicates that the economic rotation length of these species is greater than 60 years. The medium-lived NWPS species, 23% of the homegarden trees, are estimated to have half the economic rotation length of the long-lived NWPS species, or 30 years. The timber species, 20% of the homegarden trees, are estimated to have a rotation age of 15 years, or half the maximum rotation age reported by farmers and forestry staff in Lampung (see Table 4). Considering these species classes and rotation lengths, the time-averaged above-ground C stock for Lampung homegardens is estimated to be 56.5 Mg C ha^{-1} (see Example 1 in Table 5). This is 54.3 Mg ha^{-1} more (24.7 times greater) than the time-averaged C stock reported for *Imperata*-cassava systems (Palm et al., 1999). This projection suggests that increased C storage can be achieved by converting *Imperata* or fallow agricultural land to homegarden, or similar smallholder agroforestry systems.

Table 4. Typical harvest age of timber species in the study area (Roshetko and Purnomosidhi, 1998).

Botanical name	Rotation	Source
Acacia mangium	7–10 years	Forest office staff
Alstonia scolaris	15 years,	Farmers
	15–20 years	Forest office staff
Eucalyptus sp.	7–10 years	Forest office staff
Gmelina arborea	7–10 years	Forest office staff
Paraserianthes falcataria	7–8 years,	Farmers
	7–10 years	Forest office staff
Peronama canescens	20 years,	Farmers
	30 years	Forest office staff
Swietenia macrophylla	30 years	Forest office staff
Tectona grandis	+20 years,	Farmers
	30 years	Forest office staff

Realizing that the projection in Example 1 is based on assumptions regarding the species classes and rotation lengths that might be inaccurate, it is wise to recalculate the time-averaged above-ground C stock of homegarden systems with more conservative assumptions. If the maximum length of each rotation class is reduced by one-third, the time-averaged above-ground C stock for the homegarden systems is 38.8 Mg C ha^{-1}. If the maximum rotation lengths are reduced by half, the time-averaged above-ground C stock is still 30.2 Mg C ha^{-1} (see Examples 2 and 3 in Table 5). These conservative estimates are both greater than the time-averaged above-ground C stock reported for *Imperata*-cassava systems, by 36.6 Mg ha^{-1} (16.6 times greater) and 28.0 Mg ha^{-1} (12.7 times greater), respectively. This indicates that even under unfavorable assumptions homegardens still sequester C at much greater rate than *Imperata*-cassava systems.

It is also possible that the first projection is too conservative because it is based on current C stocks of 35.3 Mg of above-ground C ha^{-1}, the averaged C stock of 19 homegardens. It is fair to argue that this is an unreasonably low starting point from which to project future C stock levels, when the purpose of the projection is to target smallholder agroforestry systems for increased C storage. Seven of the homegardens surveyed in the study have current above-ground C stocks ranging from 45.4 to 84.0 Mg C ha^{-1}, with an average of 59.0 Mg ha^{-1}. By using this sub-set of seven homegardens and assuming a higher percentage of species with longer rotation lengths, projected C stock levels of homegarden will increase over that in Example 1. Table 6

Table 5. Projection of time-averaged above-ground carbon stock of Lampung homegardens, assuming current species component and projected linear growth for three rotation classes for various rotation ages.

Rotation age class and other carbon pools	% of HGS[1]	Current above-ground C stock (Mg ha^{-1})	Maximum/ current age (years)	Maximum above-ground C stock (Mg ha^{-1})	Time-averaged above-ground C stock (Mg ha^{-1})
Example 1					
Long-lived NWPS[2] species	57	18.4	60 / 13	84.9	42.5
Medium-lived NWPS species	23	7.4	30 / 13	17.1	8.5
Timber species	20	9.5	15 / 13	11.0	5.5
Total	100	35.3	–	113.0	56.5
Example 2: reduce the rotation age of example 1 by one-third					
Long-lived NWPS species	57	18.4	40 / 13	56.6	28.3
Medium-lived NWPS species	23	7.4	20 / 13	11.4	5.7
Timber species	20	9.5	13 / 13[3]	9.5	4.8
Total	100	35.3	–	77.5	38.8
Example 3: reduce the rotation age of example 1 by half					
Long-lived NWPS species	57	18.4	30 / 13	42.5	21.2
Medium-lived NWPS species	23	7.4	15 / 13	8.5	4.3
Timber species	20	9.5	13 / 13[3]	9.5	4.8
Total	100	35.3	–	60.5	30.2

[1] HGS, homegarden systems.
[2] NWPS, non-wood forest products and services species.
[3] For timber species the rotation age is reduced to the current age.

provides two examples of projected time-averaged C stocks for homegardens under these two assumptions, namely a higher current above-ground C stock (59.0 Mg ha^{-1}) and species with longer rotations. These examples estimate time-averaged above-ground C stocks of 122.5 and 104.4 Mg C ha^{-1}, that are 120.3 Mg (54.7 times) and 102.2 Mg (46.5 times) greater than the time-averaged C stock of *Imperata*–cassava systems. We feel that these projections are fair estimations, as they are similar to above-ground C stock levels of 60-year-old community forests, 114–123 Mg C ha^{-1}, assuming above-ground C is 65–70% of total C (Tomich et al., 1998). To achieve high C stocks, the smallholders should develop systems that maintain high tree density, contain species with long rotation lengths and manage the system for long rotations. It would also be beneficial to limit the number of low-biomass species such as coconuts and bananas. This last consideration must be balanced with the livelihood and market objectives of the smallholders' management plan, but clear opportunities exist for smallholders to select management practices that favor increased C stocks. However, smallholders are not going to select these management practices solely to provide society with C sequestration services. Incentive mechanisms need to be developed to assure that smallholders will benefit from selecting management practices that favor higher C stocks.

Recent studies indicate there are over 220,000 ha of *Imperata* land in Lampung, 8.5 million across Indonesia, and 35 million throughout Asia (Garrity et al., 1997). These ecosystems are prone to burn and generally underutilized, both biologically and economically. They represent a vast underutilized land resource, part of which could be used to establish tree-farming systems to meet smallholders' household and income needs (Tomich et al., 1997), while also making a significant contribution to the regional C budget. In conversations with the authors, smallholders in Pakuan Ratu show a clear interest in expanding their tree-farming activities, particularly on fallow agricultural land, which is quickly invaded by *Imperata*. Farmers are interested in tree farming for the following reasons: (1) to diversify and intensify their farming systems and income streams; (2) to develop private tree resources to meet household and market demand; and (3) to make better use of their limited labor and financial capital (Roshetko and Purnomosidhi, 1998). However, despite their experience with homegardens and interest in tree farming, most smallholders have little to no experience with intensive tree planting or management systems. Although the activities of individual farmers have been successful, many local tree-farming activities suffer from poor species selection and the use of inferior germplasm. This is evident in the species composition recorded during this study. Sengon putih (*Paraserianthes falcataria*) and rambutan (*Nephelium lappaceum*) account for 21% of the trees, both living and dead, in the homegardens. However, both species appear to be poorly adapted to Pakuan Ratu. Twenty-five percent of the rambutan surveyed were dead, a result of their sensitivity to drought; this is 33% of the total dead trees surveyed. Sengon putih shows inconsistent performance both biophysically and economically. The preference for these species results from farmers' experience with the species on other sites in Lampung and Java, or their promotion by government programs. Farmers in the study area are quick to adapt technology that serves their needs; however, they lack reliable technical input regarding species selection, germplasm quality, tree propagation, tree-farm management and markets for tree products. Additionally, farmers' initiatives to plant trees may be stifled by government regulations that are perceived to limit farmers' tree utilization rights. A team of socio-economic, forestry, horticultural and livestock specialists visiting the study area determined that smallholders' keen interest in tree farming and the productivity of those tree systems would benefit greatly from assistance in the form of technical information, resources and consultation (Gintings et al., 1996).

Smallholder agroforestry systems are not, at present, a common C sequestration project type, because working with a large number of independent smallholder farmers presents many unique challenges for C investment. However, there are large numbers of smallholder farmers in the tropics and vast areas of degraded land in need of rehabilitation. Agroforestation of these

Table 6. Projection of time-averaged above-ground carbon stock for homegardens, assuming a current above-ground carbon stock level of 59.0 Mg C ha^{-1} and species components with longer rotation lengths.

Species component	% of HGS[1]	Current C above-ground stock (Mg ha^{-1})	Maximum / current age (years)	Maximum C above-ground stock (Mg ha^{-1})	Time-averaged above-ground C stock (Mg ha^{-1})
Example 4					
Long-lived NWPS[2] species	80	47.2	60 / 13	217.9	108.9
Medium-lived NWPS species	20	11.8	30 / 13	27.2	13.6
Total	100	59.0	–	245.1	122.5
Example 5					
Long-lived NWPS species	60	35.4	60 / 13	163.4	81.7
Medium-lived NWPS species	20	11.8	30 / 13	27.2	13.6
Timber species	20	11.8	20 / 13	18.2	9.1
Total	100	59.0	–	208.8	104.4

[1] HGS, homegarden systems.
[2] NWPS, non-wood forest products and services species.

areas would sequester C and could prevent further deforestation by providing on-farm sources of trees (Sanchez, 1994; Schroeder, 1994). Simultaneously, the development of smallholder tree-farming systems would improve the livelihood of farm families. The CDM of the Kyoto Protocol will provide opportunity for the development of projects that have the dual objectives of C sequestration and sustainable social development. The structure of CDM projects might be unilateral (activities undertaken by a single entity who would then sell the CERs on the international market), bilateral or multilateral. The multilateral approach is likely to be the most effective for involving communities of smallholder farmers and having a strong focus on sustainable development (CIFOR, 2001). Projects that involve numerous smallholders will have higher transaction costs, justified as the added costs required to achieve more equity distribution of project benefits. The design and implementation of C sequestration projects that target smallholder agroforestry systems for sustainable development are just beginning to be addressed. Research issues related to smallholder systems as CDM projects were identified at a recent international workshop hosted by the Center for International Forestry Research (CIFOR) in Bogor, Indonesia (CIFOR, 2001). Some of the key questions identified include: How can C sequestration projects involving smallholders be made more attractive to investors? How can the transfer payments be disbursed efficiently and equitably to large numbers of households? What combination of financial and non-financial payments are appropriate? How can contractual agreements with communities for C (and other environmental) services be created, enforced and verified in a cost-effective way? What type of institutional arrangements and enabling regulatory environment are needed to facilitate transfer payments to communities? These questions are the focus of a new project being implemented by the International Fund for Agricultural Development (IFAD) and the International Centre for Research in Agroforestry (ICRAF), with collaboration from Winrock International, the World Resources Institute and CIFOR (ICRAF, 2000). The project seeks to develop models for rewarding rural communities for the environmental services they provide to society. The resulting models will help develop guidelines for the design and implementation of smallholder-focused C sequestration projects.

In the case of Lampung, a C sequestration project could provide farmers with financial incentive as well as the technical information, inputs and expert consultation required to convert *Imperata* or other underutilized lands to productive tree-farming systems. Such a project would be a wise investment in social, political, economic and environmental terms. We would expect 'leakage'—the loss of C, primarily as woody biomass, in outside areas due to changes in land-use practices resulting from C investment activities at a project site—to be minimal or non-existent. The conversion of some *Imperata* lands is not likely to greatly alter local land-use practices that would result in losses of C elsewhere, particularly when abundant *Imperata* lands remain. Nor are negative market impacts expected to result from an increased supply of smallholder-produced tree products, because the current demand for both timber and fruit in Lampung exceeds local production (Suparman, 2000; Yuliyanti, 2000). More problematic is the question of 'additionality'—assuring that C credits accrued to a C sequestration project are 'additional' to those that would occur without the project. As previously stated, the initiative of farmers to expand tree-farming systems is hampered by technical problems and other issues. However, some farmers do plant trees. Determining accurately the amount of 'additional' carbon sequestered by project incentives will rely on the establishment of quantifiable baseline data. However, the development of baseline data in smallholder communities could be difficult because of the plethora of landowners, their objectives, land-use systems and other factors. The development of guidelines for the establishment of quantifiable baseline data is an important issue that has not yet been addressed.

Although at this point there are more questions than answers concerning the implementation of smallholder-focused C sequestration projects, we believe such projects are highly desirable from both a C investment and sustainable development point of view. Agroforestation of *Imperata*, or other low-biomass lands, would increase a region's C budget and provide many socio-economic benefits to the people living in the region. Mechanisms should be explored that facilitate the implementation of smallholder-focused projects.

Conclusion

Individual smallholder agroforestry systems are of limited size and by themselves store small amounts of C. However, on a per area basis, homegardens and other smallholder agroforestry systems accumulate significant amounts of C, equaling the amount of C stored in other tree-based systems—including primary or secondary forests—over similar time periods. However, smallholder systems greatly exceed the amount of C stored by *Imperata* grasslands or agricultural fallow land, which are extensive in many parts of Indonesia and of low productivity. Thus, aggregate smallholder systems offer the potential for increased C sequestration via tree establishment on *Imperata* and other degraded lands. These systems could be established by farmers to meet both their household production and income generation needs, and simultaneously contribute to the reduction of greenhouse gases. Governments are generally supportive of tree-planting efforts, as a means of achieving conservation, reforestation and watershed protection objectives, as well as improving the livelihoods of smallholder farm families. We suggest that it is timely and appropriate to explore the CDM and other mechanisms by which communities or consortia of smallholder farmers can access international C investment funds to convert low-biomass lands, such as *Imperata*–cassava lands, to productive tree-based systems which contain much higher C stocks.

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Macroeconomic impacts of reducing greenhouse gas emissions from Canadian agriculture

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Abstract. *Canada's commitment under the Kyoto Protocol is to reduce its greenhouse gas (GHG) emissions by 6% of its 1990 levels. Each industrial sector is investigating alternative technologies, production and management practices that can decrease their GHG emissions. The macroeconomic impacts of four mitigation strategies to reduce GHG emissions from Canada's agriculture sectors were measured using an input–output model. The size of the GHG reduction from each mitigation strategy depended on whether agricultural soils were included as a carbon (C) sink. Including agricultural soils as a C sink impacts on the absolute amount of GHG emissions that must be reduced and the relative importance of the various mitigation strategies. This will be a key factor in policy development. Only one strategy, improving forage quality by 15%, had positive macroeconomic impacts in all situations. It was projected that this strategy would increase industrial output by $106.97 M (M = million; all $ Canadian), gross domestic product at factor cost (GDP) by $45.51 M and employment by 689 jobs. This strategy decreased GHG emissions by 0.07% below the 'business as usual' (BAU) situation when sinks were included. Increasing the adoption of zero-till farming had a positive macroeconomic impact only when the industrial sector effects were included. However, when household and industrial-sector impacts were combined, the results were decreases in industrial output of $286.90 M, GDP of $55.98 M and employment by 769 jobs. The mitigation strategy decreased GHG emissions by 3.06% below the BAU situation when sinks were included in the estimate. Improved soil nutrient management through more efficient use of N fertilizer had a negative net impact on the economy. This mitigation strategy had a direct impact on the agriculture and the fertilizer sectors, resulting in net decreases in industrial output of $70.76 M, GDP of $43.38 M and employment of 518 jobs. It was estimated that this mitigation strategy would decrease GHG emissions by 1.37% below the BAU situation. The last mitigation strategy was a permanent plant cover program. This generated the largest negative impact on the economy. It was projected to decrease industrial output by $1192.63 M, GDP by $392.17 M and employment by 6155 jobs. The strategy decreased GHG emissions by 1.73% below the BAU situation.*

Key words: alternative production practices, Canadian economy, carbon sequestration, carbon sink, greenhouse gas mitigation, input–output model, soil carbon, zero till

Introduction

Under the terms of the Kyoto Protocol, Canada agreed to reduce its greenhouse gas (GHG) emissions by 6% below its 1990 levels (UNFCCC, 1997). In response, the government of Canada established the Climate Change Secretariat which, in turn, established 16 Tables tasked with developing various mitigation strategies to reduce GHG emissions. The federal and provincial ministries of energy and environment will use the output from these Tables to develop a National Implementation Strategy.

The Agriculture and Agri-Food Climate Change Table (AAFCCT) was established specifically to identify and analyze potential mitigation strategies for the agriculture sectors and food processing sectors of the economy. Only the mitigation strategies for the agriculture sectors will be included in this analysis. Mitigation strategies for the agriculture sectors included changes in technology and production and management practices that would decrease the amount of GHG emissions relative to current production and management practices. These alternative production and management practices were assessed in terms of their environmental, economic, social and health impacts.

Agriculture production activities in Canada accounted for approximately 9.5% of the national GHG emissions in 1990 (Neitzert et al., 1999). One unique characteristic of the agricultural sector is that GHG emissions are primarily from biological and physical sources. This is different from other industrial sectors where energy consumption is the main source of GHG emissions. As a result, agriculture's GHG emissions profile has a higher proportion of N_2O (61%) and CH_4 (38%) relative to CO_2 (1%) as compared to other industrial sectors (AAFCCT, 2000).

Having biological and natural sources as the main contributors of GHG emissions created two problems in the development and analysis of appropriate mitigation strategies

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for this sector. The first was the lack of scientific understanding of the complex natural biological and physical processes of emission. In particular, how do GHG emissions vary in a spatial context due to differences in soil type, management practices and climate conditions? As a result, changes in GHG emissions were based on the best available information; however, more scientific analysis is needed to improve the reliability of the estimates. Second, under the current terms of the Kyoto Protocol, agricultural soils are not recognized as C sinks. Including agricultural soils as C sinks has been controversial during recent negotiation sessions and will have a major impact on the strategies that are adopted for GHG emission reduction.

The Agriculture and Agri-Food Climate Change Table (2000) followed a three-phase process for identifying and analyzing the mitigation strategies. Phase 1 was to identify soil, crop, animal and agroforestry practices that could potentially reduce GHG emissions. In this phase, scientists from various disciplines identified potential mitigation strategies that could be applied. The selection of the mitigation strategies to be analyzed in Phase 2 was determined by the AAFCCT. Phase 2 used a series of models to measure the environmental, social, economic and health impacts of selected mitigation strategies. Phase 3 was the generation of an option paper by AAFCCT (2000).

The objective of this paper was to measure the macroeconomic impacts on the Canadian economy of adopting various mitigation strategies to reduce GHG emissions. These projections include not only the direct impact on the agriculture sectors of adopting these alternative production and management practices, but also the impact on other industrial and household sectors in the economy as they adjust to changes in the agriculture sectors. This information is important for decision-makers because they must take into account the broader economic impacts of their policy decisions regarding GHG emission reductions. Macroeconomic impacts measure the net effect on the economy, either positive or negative, and identify those industrial sectors most affected by each mitigation strategy. For agricultural producers, this information is useful because it quantifies the impact of their decisions, i.e., choosing alternative production and management practices, on other sectors in the economy. The specific objective of this paper is to measure the changes in industrial output, gross domestic product at factor cost (GDP) and employment that are associated with four mitigation strategies for reducing GHG emissions in the agriculture sectors. These strategies are: (1) improved soil nutrient management; (2) increased use of zero till; (3) improved forage quality; and (4) permanent cover program.

Model Selection and Mathematical Derivation

An input–output modeling approach was chosen to measure the macroeconomic impacts of the mitigation strategies on the economy, for several reasons. First, the input–output framework is a system-based modeling approach that incorporates the interrelationships within and between industrial and household sectors. This system-based approach attempts to capture all of these relationships and thus can measure the impact of a change in one sector on all other sectors of the economy. Second, Statistics Canada has a long tradition of constructing reliable input–output tables for the Canadian economy that would be the basis of the models used in this analysis. Finally, these input–output tables can be modified to provide additional details for the agriculture and household sectors.

The Canadian input–output model, which has been described elsewhere (Miller and Blair, 1985; Poole, 1995; Thomassin and Andison, 1987; Thomassin and Baker, 2000; Thomassin et al., 1992), is based on an accounting framework where the number of industries does not equal the number of commodities. The advantage of this framework is that each industrial sector can produce more than one commodity. All monetary transactions are accounted for in one of five matrices: intermediate demand matrix (U), primary inputs (YI), a matrix of commodity outputs (V), a final demand matrix (F), and a primary inputs matrix going into final demand categories (YF). A number of economic relationships are contained in these matrices. The first is that the value of industrial output, g, is equal to the summation of the value of each commodity produced by each industrial sector, Vi as

$$g = Vi \tag{1}$$

where i is a vector of 1s.

The second relationship defines the disposition of commodities to categories of demand. The total demand for commodities in the economy, q, is equal to the intermediate demand for commodities, U, plus the final demand for commodities, F:

$$q = Ui + Fi \tag{2}$$

The input–output model can be estimated using these two relationships, and making assumptions concerning industrial technology and the market share of industrial sectors. In terms of industrial technology, it was assumed that the current inputs used by each industry were proportional to the output produced by that sector:

$$U = B\hat{g} \tag{3}$$

where $\hat{}$ indicates a diagonal matrix and B, a matrix of technical coefficients.

The market share assumption is that the demand for commodities produced in the economy is allocated to industrial sectors in fixed market shares:

$$V = D\hat{q} \tag{4}$$

where D is a matrix of market share coefficients.

Starting with Equation 1 and substituting with Equations 4, 2 and 3 provides an estimate of the input–output model, that is,

$$g = (I - DB)^{-1}Dfi \tag{5}$$

where I is the identity matrix.

Equation 5 does not take into account leakages out of the economy that result from imports, withdrawals from inventories or changes in government production. To take these into

account, it was assumed that the leakages were in fixed proportion to the domestic commodity demand. This assumption can be incorporated into Equation 5 as:

$$g = (I - D(I - \hat{\mu} - \hat{\eta} - \hat{\alpha})B)^{-1}D[(I - \hat{\mu} - \hat{\eta} - \hat{\alpha})f + (I - \hat{\mu})E + (I - \hat{\eta} - \hat{\alpha})X] \quad (6)$$

where:

$\hat{\mu}$ = a diagonal matrix of imports to commodity use,

E = a vector of re-exports,

f = a vector of final demand, excluding exports, re-exports, imports, government production and withdrawals from inventory,

$\hat{\eta}$ = a diagonal matrix of inventory withdrawals to commodity use,

X = a vector of commodity exports, and

$\hat{\alpha}$ = a diagonal matrix of government production to commodity use.

From Equation 6, $[I - D(I - \hat{\mu} - \hat{\eta} - \hat{\alpha})B]^{-1}$ is the inverse matrix. Post-multiplying the inverse matrix by D, provides an estimate of the impact matrix, $[I - D(I - \hat{\mu} - \hat{\eta} - \hat{\alpha})B]^{-1}D$. The impact matrix can measure the direct plus indirect change in industrial output, g, required to satisfy a change in the final demand for commodities, q.

The impact matrix estimated above is for an open input–output model that estimates the direct plus indirect impacts on the economy. This model measures the inter-industry relationships in the economy. Household behavior can be incorporated into the model by endogenizing household expenditure and revenue patterns. The technique used to close the model to households was similar to that used by Cloutier and Thomassin (1994). Ten household sectors, based on their income level, were incorporated into the model. Each had its own expenditure and revenue pattern. The closed version of the model measures the direct, indirect and induced change in industrial output required to satisfy a change in the final demand for commodities.

Macroeconomic Analysis

The open version of the Agriculture and Agri-Food Canada (AAFC) Input–Output Model includes 13 agricultural sectors, 16 food- and beverage-processing sectors and 46 other industrial sectors (see Appendix 1 for a list of the agriculture, food- and beverage-processing and household sectors included in the model). The model quantifies the impact on the economy by measuring the backward linkages between sectors in the economy. An example of a backward linkage would be: if agricultural producers increase the quantity of wheat (*Triticum aestivum* L.) produced, they would have to purchase inputs (fertilizer, seed, fuel, etc.) from other sectors in the economy. These other sectors, in turn, would have to purchase inputs in order to produce the products needed by the agriculture sectors. As a result, the model will measure the change in industrial output for all industrial sectors in the economy from a change in the demand for commodities produced by the agriculture sectors. The impacts measured by the model include changes in industrial output, GDP and employment.

The closed version of the model estimates the direct, indirect and induced impacts, where the induced impacts are those that result from a change in household spending. For example, if agricultural producers change their production practices, resulting in a decrease in net income, this would have a negative impact on the economy because producers would have less to spend on household goods and services. If producers have less income, the demand for goods and services would decrease, which would result in less goods and services produced. With fewer goods and services, there is a decrease in the demand for the inputs needed to produce these goods and services.

Input–output models measure the macroeconomic impacts on the economy by estimating the industrial output required to satisfy a change in the final demand for commodities. The changes in final demand for commodities were estimated with the Canadian Economic and Emissions Model for Agriculture (CEEMA). CEEMA contains two components, the first is an economic optimizing model and the second is a GHG emission model (AAFC, 1999b, 1999c). The economic optimization model of CEEMA was used to estimate the output configuration of the business as usual (BAU) situation. Each mitigation strategy was then introduced into the model, which generated a new configuration of commodity outputs for the agriculture sectors. The difference in output configuration between the BAU situation and the mitigation strategy provided the change in final demand for the scenario. Changes could be either positive or negative for individual commodity outputs. Farm household income changes were estimated by the change in net margins between the BAU situation and the mitigation strategy. Net margins are defined as the difference between revenue and costs.

Scenario development

From phase one of the process, 11 mitigation strategies were identified as possible means of reducing GHG emissions from the agriculture sectors. GHG emissions were estimated using two techniques. The first used the Intergovernmental Panel on Climate Change (IPCC) methodology that estimates the GHG emissions from direct production of crops and animals and some indirect emissions. The second used the GHG emissions model in CEEMA. The CEEMA emission estimates are based on the CENTURY model coefficients modified for Canada (Junkins et al., 2000). They include the IPCC impacts, on-farm energy use and C sequestered from agricultural sinks. These latter two elements are not included in the IPCC estimates.

GHG emissions for 1990 were estimated with CEEMA using the 1990 baseline, which was based on the 1991 Census of Agriculture estimates of livestock numbers, area planted and tillage practice. This was done because Canada's commitment in the Kyoto Protocol is based on decreasing its emissions from this reference year (i.e., 6% below 1990 levels). Total cropland area included 45 M (M = million) ha of which 35 M ha were cropped and 10 M ha were hayland or improved pasture. With the 1990 baseline, it was estimated that the agriculture sectors produced 57.6 MT (million tonnes = megatonne) of CO_2 equivalents using

Table 1. Scenario descriptions and GHG emissions.[1]

Scenario description	GHG impact: IPCC[2]/inventory (% from BAU[3])	GHG impact[4]: total primary agriculture[5] (% from BAU)
Soil nutrient management – improved N fertilizer management	–1.47	–1.37
Increased utilization of zero till	0.61	–3.06
Permanent cover program	–0.91	–1.73
Improved forage quality	–0.36	–0.07

[1]Source: Junkins et al. (2000).
[2]IPCC is the Intergovernmental Panel on Climate Change.
[3]BAU is the Business As Usual scenario.
[4]GHG impacts were estimated with CENTURY model coefficients adopted for Canadian conditions (Junkins et al., 2000).
[5]Total primary agriculture impacts include the IPCC/Inventory level plus on-farm use of fossil fuels and sinks (C sequestered by soil).

the IPCC method of calculation (Junkins et al., 2000). Livestock production (N_2O and CH_4) accounted for the largest proportion of these emissions. Taking into account sinks and on-farm fossil fuel use, the GHG emission estimates increased to 65.67 MT of CO_2 equivalents (Junkins et al., 2000). These estimates were comparable to other studies on agricultural GHG emissions for Canada (AAFC, 1999c).

Canada expects to satisfy its Kyoto Protocol commitment sometime during a target period between 2008 and 2012. Thus, any increase in GHG emissions that occurs between 1990 and 2010 must be accounted for and taken into consideration as part of the Kyoto Protocol commitment. In order to project the level of GHG emissions that would occur during this period, a BAU projection was made to 2010. The 2010 BAU projection was based on the Medium Term Policy Baseline (AAFC, 1999a). Crop, tillage practices and livestock projections were based on the historical trends from the 1981, 1986, 1991 and 1996 Census of Agriculture and the Medium Term Policy Baseline (Junkins et al., 2000). In the 2010 BAU projections, the amount of summerfallow decreased by 2.8 M ha from 1990 values, while the amount of cropland cultivated with zero-till management increased to 30% on average in Alberta, Saskatchewan and Manitoba. The GHG emissions estimated with the 2010 BAU scenario were 65 MT of CO_2 equivalents using the IPCC method (Junkins et al., 2000). This was an increase of 13% above 1990 levels. Using this methodology, the agriculture sectors would have to decrease GHG emissions by 19% to satisfy their Kyoto Protocol commitment.

Taking into account sinks and on-farm fossil fuel use, the estimated GHG emissions increased to 67.3 MT of CO_2 equivalents (Junkins et al., 2000), a 2.4% increase over the 1990 baseline. The difference in the two estimated values is the result of including sinks, and their C sequestering credit, into the GHG emission estimates. Carbon is credited to agricultural soils because of an increase in zero- and minimum-tillage practices and a decrease in the amount of summerfallow. Using this method to estimate GHG emissions, the agriculture sectors would have to decrease their GHG output by 8.4%.

The macroeconomic impacts reported in this paper are for four mitigation strategies: soil nutrient management, increased utilization of zero till, improved forage quality and a permanent cover program (Table 1). The soil nutrient management, improved forage quality and permanent cover program strategies were selected because they resulted in a decrease in GHG emissions using both estimation methods. The zero-till strategy was selected because it provided a large decrease in GHG emissions when sinks and on-farm energy use are included.

Soil nutrient management

The soil nutrient management mitigation strategy includes changes in management practices that would improve the efficiency of N use by crops by better matching N availability with crop requirements. These management strategies include the use of soil tests for selected crops in eastern Canada to improve fertilizer management and the elimination of fall fertilizer and manure applications in Alberta, Saskatchewan and Manitoba. These soil nutrient management methods would decrease the N_2O flux into the atmosphere and thus, N_2O emissions, a major component of GHG emissions in agriculture. Soil tests should decrease the amount of N fertilizer applied to a crop and may reduce crop input costs. Changing fertilizer applications from the fall to the spring in the provinces of Manitoba, Saskatchewan and Alberta, increases N use efficiency and, thus, reduces the quantity of fertilizer required (Thomsen Corporation, 1999).

This mitigation strategy included the following changes in fertilizer applications: a 20 kg ha^{-1} reduction for cereals and a 30 kg ha^{-1} reduction on 50% of the potato (*Solanum tuberosum*) hectarage in the Atlantic Region; no change in Quebec; a 30 kg ha^{-1} reduction in Ontario; an average 2.1 kg ha^{-1} reduction in fertilizer use in Manitoba, Saskatchewan, Alberta and the Peace River District; and a 50 kg ha^{-1} reduction on 55% of the cropland in the interior and lower mainland British Columbia (Junkins et al., 2000). With this scenario it was assumed that crop yields would not be impacted by this reduction in fertilizer used and that producers would adopt these rates by the year 2010.

The direct impacts of this scenario were allocated to changes in the demand for commodities. These fell into two industrial sector categories. The first was the decrease in the demand for fertilizer, which had a negative impact on the fertilizer industry.

Table 2. Macroeconomic impact of the decrease in the demand for fertilizer with the soil nutrient management scenario.

Impact	Industrial output (in million $ cdn)	GDP at factor cost (in million $ cdn)	Paid employment (jobs)
Direct effect	–79.09	–36.28	–391
Direct + indirect effect	–142.31	–63.44	–793
Direct + indirect + induced effect	–272.90	–95.28	–1295

Table 3. Macroeconomic impact of changes to the agriculture sectors as a result of the soil nutrient management scenario.

Impact	Industrial output (in million $ cdn)	GDP at factor cost (in million $ cdn)	Paid employment (jobs)
Direct effect	14.11	3.68	8
Direct + indirect effect	36.41	12.47	139
Direct + indirect + induced effect	202.14	51.90	777

Table 4. Soil nutrient management scenario: net impacts.

Impact	Industrial output (in million $ cdn)	GDP at factor cost (in million $ cdn)	Paid employment (jobs)
Direct effect	–64.98	–32.6	–383
Direct + indirect effect	–105.91	–50.97	–654
Direct + indirect + induced effect	–70.76	–43.38	–518

The other was the agriculture sectors which adjusted their output with both positive and negative changes in commodity outputs.

Better management of N fertilizer results in a decrease in the demand for fertilizer by the agriculture sectors. The direct, indirect and induced impacts of this change were decreases in industrial output of $272.90 M (all measures of industrial output and GDP are in Canadian dollars), in GDP of $95.28 M, and in employment of 1295 jobs (Table 2). The industrial sectors most affected were the mining and the chemical and chemical products sectors. These two sectors had a decrease in industrial output of $40.15 M and $21.35 M, respectively. When taking into account the direct, indirect and induced effects, the five sectors most affected by the decrease in fertilizer demand were: mining, households with greater than $60,000 income, chemical and chemical products, finance and real estate, and wholesale trade. These account for 46% of the decrease in industrial output, 48% of the decrease in GDP and 37% of the decrease in employment.

The macroeconomic impacts of changes in the agriculture sectors are shown in Table 3. The direct impact of this scenario was positive, but small. The total impact from changes in the agriculture sectors was increases in industrial output of $202.14 M, in GDP of $51.90 M, and in employment of 777 jobs. The sectors most affected by this change in output were: grains and oilseeds, wheat, meat and meat products, cattle and transportation. These account for 66% of the increase in industrial output, 67% of the increase in GDP and 89% of the increase in employment when the direct plus indirect effects were taken into account. Sectors that were adversely affected by this scenario included fruit, vegetables and potatoes.

The largest impacts occur in the direct, indirect and induced effects because of the increase in net margins that are generated with improved management of soil nutrients. These were predicted to increase by $54.8 M. The increase was divided between the crop ($63.1 M) and livestock sectors (–$8.4 M) (Junkins et al., 2000). Net margins for crops increased in the West but decreased in the East, while the opposite occurred for the livestock sectors (AAFC, 1999d). The sectors most affected by this scenario were: households with greater than $60,000 income, grains and oilseeds, finance and real estate, households with $50,000–$59,999 income, and households with $40,000–$49,999 income. These sectors accounted for 52% of the change in industrial output, 24% of the GDP and 16% of the increased employment.

The net impact of this scenario is given in Table 4. In this case, the economy was adversely affected by the changes in industrial output (–$70.76 M), which decreased GDP by $43.38 M and decreased employment by 518 jobs. These results indicate that the positive impacts to the agriculture sectors do not offset the decrease in economic activity in the fertilizer industry.

A number of factors should be taken into consideration when analyzing these results. First, it was assumed that the change in timing of fertilizer applications from the fall to the spring in Alberta, Saskatchewan and Manitoba would be easily undertaken and without cost. This may not be the case. There could be additional costs of on-farm fertilizer storage requirements, timing of field operations in the spring, and logistical, storage and inventory costs for the fertilizer sector. To alleviate some of these problems, additional capital investments would have to be

Table 5. Macroeconomic impacts of the increased utilization of zero-till practices.

Impact	Industrial output (in million $ cdn)	GDP at factor cost (in million $ cdn)	Paid employment (jobs)
Direct effect	26.96	13.87	382
Direct + indirect effect	50.21	23.43	531
Direct + indirect + induced effect	–286.90	–55.98	–769

made. These additional costs would decrease the positive impacts to the agriculture sectors.

It was also assumed that there would be no change in fertilizer use rate in Quebec because of a lack of reliable information for this province. However, one would expect a decrease in fertilizer use in Quebec if adequate soil testing was done. Such a program would further reduce the GHG emissions associated with this mitigation strategy.

Comparing this mitigation strategy with the 2010 BAU situation, it has the smallest impact on direct industrial output ($14.11 M) of the four scenarios investigated. The change in quantity and mix of outputs of the agriculture sectors under this scenario was approximately 0.04% higher than the 2010 BAU case. This mitigation strategy does not require a substantial shift in industrial output from the BAU situation.

Finally, taking into account the broader economic impacts of this mitigation strategy has important public policy implications. The positive direct impacts to the agriculture sectors are offset by the negative impacts in other industrial sectors, thus transferring the cost of this mitigation strategy to other sectors. This information is important for public policy makers as they attempt to identify mitigation strategies with the lowest economic costs and the distribution of those costs to industrial sectors in the economy.

Increased utilization of zero till

Zero till is a management practice that conserves crop residues by reducing soil disturbance and thus improves its C sink potential. Adopting zero till does, however, impact crop production inputs, with an increase in herbicide use for weed control but a decrease in machinery and fuel use. With this mitigation strategy it was assumed that the amount of zero till would increase to 52% of the cropland in Alberta, Saskatchewan, Manitoba and the Peace River region. This is an increase of 21% over the BAU projection. The additional hectares going into zero till production would come from land that was in conventional and minimum-till practices in the BAU baseline. It was assumed that crop yields would not change with the adoption of this tillage management practice (ESG International, 1999).

The impacts of this mitigation strategy are given in Table 5. The CEEMA model estimated both positive and negative impacts on the agriculture sectors (Junkins et al., 2000). Positive changes in output were estimated at $132.4 M, while the negative impacts were $105.4 M. The open model estimated an increase in industrial output of $50.21 M, GDP of $23.43 M and an increase in employment of 531 jobs (Table 5). The sectors with the largest net impact were: wholesale trade, vegetables, fruit, potatoes, and meat and meat products. Industrial sectors that were negatively impacted by this scenario included: grains and oilseeds, dairy, cattle, poultry and eggs, and combination livestock.

However, this scenario reduced net margins to producer by $151.7 M (Junkins et al., 2000). Net margins for the crop sectors decreased in all regions except British Columbia, while for livestock net margins increased in all regions except Saskatchewan.

The overall impact of this mitigation strategy on the economy was negative. The closed model projected decreases in industrial output of $286.90 M, in GDP of $55.98 M and in employment of 769 jobs. These results suggest that the input suppliers to the agricultural sectors would be better off with this strategy; however, these gains would be offset by the negative impact in other industrial sectors.

Two important factors to consider when evaluating this mitigation strategy are the assumptions related to crop yields and the adoption rate. If it were assumed that yields would decrease, this would have a negative impact on net margins from crops and would exacerbate the negative induced effect on farm income. As a result, the negative impact on the economy would be larger.

The second assumption that needs to be considered is the amount of land allocated to zero till, i.e., 52% of all cropland. This was a substantial increase from the BAU projection. If this level of zero till is not reached by 2010, it will impact the GHG emission reductions predicted in Table 1. Public policy decision-makers should take this into consideration when selecting mitigation strategies.

The adoption of zero till by producers would require the purchase of additional farm equipment. This impact was not considered in this analysis. It would increase costs to producers but have a positive impact on the farm machinery sector.

The economic impacts measured for this mitigation strategy change from positive to negative as the induced effect is included (Table 5) and highlights the relationship between net margins and industrial sector output. As the decrease in net margins works its way through the economy it overshadows the positive increase in industrial output that results from this mitigation strategy. This should be taken into consideration in the public decision-making process.

Improved forage quality

This mitigation strategy was designed to improve forage quality by 15% through using better harvesting and storage methods, such as plastic wrap on hay bales. Improving forage quality decreases CH_4 emissions per unit of energy consumed by

Table 6. Macroeconomic impacts of improving forage quality by 15% throughout Canada.

Impact	Industrial output (in million $ cdn)	GDP at factor cost (in million $ cdn)	Paid employment (jobs)
Direct effect	60.19	20.58	336
Direct + indirect effect	140.38	52.54	819
Direct + indirect + induced effect	106.97	45.51	689

Table 7. Macroeconomic impacts of a permanent cover crop program.

Impact	Industrial output (in million $ cdn)	GDP at factor cost (in million $ cdn)	Paid employment (jobs)
Direct effect	–311.22	–111.81	–1841
Direct + indirect effect	–708.45	–273.35	–4298
Direct + indirect + induced effect	–1192.63	–392.17	–6155

ruminants. Appropriate tests would also be undertaken to improve feed intake efficiency. It was assumed that 20% of all beef classes would be fed improved forage and that this would result in a 12% increase in the rate of gain (Junkins et al., 2000).

Adopting this mitigation strategy had both positive ($74.0 M) and negative ($13.8 M) impacts on the agriculture sectors (AAFC, 1999d). The direct plus indirect effects were projected to increase industrial output by $140.38 M, GDP by $52.54 M and employment by 819 jobs (Table 6). The five industrial sectors that benefit most were: grains and oilseeds, wheat, meat and meat processing, transportation and wholesale trade. These sectors accounted for 51% of the increase in industrial output, 52% of the GDP and 54% of the employment.

CEEMA estimated the impact on net margins to be a negative $56.4 M. Net margins for both crop and livestock sectors were negative. All regions had a decrease in net margins in their crop sectors. For the livestock, only the Atlantic provinces had a small positive increase in net margins (Junkins et al., 2000).

As a result of the decline in net margins, the impact on the economy decreased when the induced effect was taken into account. The direct, indirect and induced impact on the economy was estimated to increase industrial output by $106.97 M, GDP by $45.51 M and employment by 689 jobs (Table 6). The same five sectors had the largest impact as with the direct plus indirect effects. These sectors accounted for 66% of the increase in industrial output, 59% of the increase in GDP and 62% of the increase in employment.

One of the assumptions important in this analysis is the rate of gain of the beef herd. If this rate is not achieved, it would decrease the positive economic effects of this mitigation strategy and have further negative impacts on the net margins of the livestock sectors.

This was the only mitigation strategy that had positive economic impacts throughout. The direct effect on the agriculture sectors' industrial output was small ($60.19 M), and was an increase of only 0.1% over the total agriculture sectors' industrial output for the 2010 BAU situation.

The only drawback to this mitigation strategy, from a public policy perspective, is its poor performance for reducing GHG emissions. The GHG emission reductions were estimated to be –0.36% and –0.07%, given the respective methodologies of the 1990 levels (Table 1). This is far below the projected required decreases of 19% or 8.4%, depending on whether or not C sinks are included, that are needed to satisfy Canada's commitment to the Kyoto Protocol. The mitigation strategy could not be the only one adopted by policy makers, but could be combined with other strategies.

Permanent cover program

With this program, 1 million ha of marginal cropland in Alberta, Saskatchewan and Manitoba would be converted to perennial forage. It would increase the C sink potential in soils by reducing the amount of soil disturbance and organic matter decomposition, and increasing the amount of biomass added to the soil. The strategy would move marginal cropland into improved pasture and hayland. It was assumed that the size of the beef herd would not change.

The CEEMA results for this scenario estimated that the positive impact on the agriculture sectors totaled $38.0 M while the negative impacts on other sectors was $349.2 M (AAFC, 1999d). The direct plus indirect effects decreased industrial output by $708.5 M, decreased GDP by $273.4 M and decreased employment by 4298 jobs (Table 7). The sectors most affected were: wheat, grains and oilseeds, transportation, wholesale trade and transportation margins. These sectors had a combined decrease in industrial output of $353.6 M, a decrease in GDP of $136.1 M, and a decrease in employment of 2195 jobs.

The producer net margins were estimated to increase by $35.9 M (AAFC, 1999d). Net margins for the livestock sectors increased in all regions, while net margins for the crop sectors increased in British Columbia, Quebec, New Brunswick, Prince Edward Island and Nova Scotia. All other regions experienced negative net margins in the crop sectors (Junkins et al., 2000).

The overall effects on the Canadian economy were decreases in industrial output of $1192.63 M, in GDP of $392.2 M, and in employment of 6155 jobs. The five most affected sectors were: households with greater than $60,000 income, wheat, grains and

oilseeds, transportation, and finance and real estate. These sectors accounted for 41% of the decrease in industrial output, 39% of the decrease in GDP and 35% of the decrease in employment.

Changing the size of the beef herd impacts the results in terms of the reductions in GHG emissions and the size of the economic effects. With an increase in the beef herd, one would expect the reduction in GHG emissions would decrease because of an increase in GHG emissions from this source. However, the increase in the beef herd would offset some of the negative economic impacts of converting marginal cropland to forage and thus increase the output and revenue for the agriculture sectors.

The negative impact of this mitigation strategy was substantially higher than for the other three strategies. It resulted in a decrease in direct industrial output of 0.9% of the 2010 BAU total industrial output for the agriculture sectors. As the decrease in industrial output works its way through the economy, the indirect and induced effects expand the negative impacts. The direct, indirect and induced effects from this mitigation strategy were 4.1 times larger than the next largest impact. As seen from Table 1, the reduction in GHG emissions from this mitigation strategy only accounts for a small portion of the required Canadian commitment.

Conclusions

The macroeconomic impacts of four mitigation strategies to decrease GHG emissions from the Canadian agricultural sectors were measured using an input–output model. Only one, an increase in forage quality by 15%, had positive impacts for the direct, indirect and induced effects. The zero-till strategy produced positive impacts at the industry level, but these became negative when changes in producer income were included. Soil nutrient management and improved forage quality strategies had negative economic impacts for the direct, indirect and induced effects. The negative impact from the improved forage quality strategy was 4.1 times larger than the next largest strategy.

The analysis illustrates the importance of including agricultural soil as a C sink in the selection of mitigation strategies. Currently, agricultural soils are not considered a C sink under the Kyoto agreement. If these are included, it would reduce the absolute amount of GHG emissions that need to be reduced, and would alter the relative importance of the various mitigation strategies to fulfill Canada's Kyoto commitment. If C sinks are excluded, soil nutrient management provides the greatest reduction (–1.47%) in GHG emissions, followed by the permanent cover program (–0.91%) and the improved forage quality strategy (–0.36%). The zero-till strategy would increase GHG emissions. Including C sinks, zero till provides the greatest reduction (–3.06%) in emissions followed by the permanent cover program (–1.73%), soil nutrient management (–0.37%) and improved forage quality (–0.07%). On this basis, policy makers should actively pursue the recognition of agricultural soils as C sinks.

None of the mitigation strategies satisfy the 19% or 8.4% projected GHG emission reductions, from 1990 levels, that would be required by the agriculture sectors to fulfill its share of Canada's Kyoto Protocol commitment. Combining all four mitigation strategies would only reduce the sectors' GHG emissions by 2.13% and 6.23% of 1990 levels, depending upon whether agricultural soil sinks are excluded or included. This result requires policy-makers to expand the number of mitigation strategies beyond the four included in this analysis if they are committed to satisfying the Kyoto commitment. One would expect that additional mitigation strategies would have higher costs to the economy in terms of decreased industrial output, and smaller percent reductions in GHG emissions.

The macroeconomic impact on the economy of each mitigation strategy was not substantial when viewed independently. However, if several mitigation strategies were adopted, the macroeconomic impacts would increase. This could result in a substantial decrease in industrial output and income, which would impact negatively on the agriculture sectors and the broader Canadian economy.

Some limitations to the analysis include the lack of scientific understanding of the effectiveness of agricultural soils as C sinks, the uncertainty concerning the recognition of agricultural soils as C sinks, and the industrial structure underlying the input–output model. The GHG emissions for the agriculture sectors were based on the best available information; however, it is recognized that the scientific understanding of these complex biological and physical processes is limited. The analysis of the mitigation strategies will be enhanced as new and more accurate information becomes available on these biological and physical processes.

The uncertainty surrounding the recognition of agricultural soils as C sinks must be resolved in order to evaluate the various mitigation strategies and to develop a comprehensive policy program. This information will have a major impact in two areas. First, it will determine the absolute reduction in GHG emissions that will be needed to satisfy Canada's Kyoto Protocol commitment. Second, it will impact the choice of technology, production and management practices that fulfill this commitment at the lowest economic cost.

The input–output model used to measure the macroeconomic impacts was based on the 1991 industrial structure of the Canadian economy. Changes in industrial structure over time will change the relationships between industrial sectors. As these change, measurements based on the old industrial structure will either under- or overestimate the actual impacts.

Areas of future research should include the following: first, better GHG accounting for different agricultural practices under different soil and climatic conditions. These improved estimates of GHG emissions can be incorporated into the policy-making framework to identify the most promising mitigating strategies. Also in need of research is the institutional structure that will provide incentives for producers to adopt the various technological, production and managerial practices to accomplish the above goals. This could include the establishment of a C trading market where GHG emissions/reductions can be purchased. Finally, the input–output model could be improved to better reflect the projected industrial structure of 2010.

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Appendix 1: Agriculture, food processing and household sectors included in the input–output model

Agriculture sectors

Dairy
Cattle
Hogs
Poultry and eggs
Livestock combinations
Wheat
Grains and oilseeds
Vegetables
Fruit
Potato
Tobacco
Greenhouse and nursery
Other farms

Food and beverage processing sectors

Meat and meat products (excluding poultry)
Poultry products
Fish products
Fruit and vegetable
Dairy products
Feed
Vegetable oil mills (excluding corn oil)
Biscuit
Bread and other bakery products
Cane and beet sugar
Miscellaneous food products
Soft drinks
Distillery products
Brewery products
Wine
Tobacco products

Household sectors (annual income)

Households with income <$10,000
Households with income between $10,000 and $14,999
Households with income between $15,000 and $19,999
Households with income between $20,000 and $24,999
Households with income between $25,000 and $29,999
Households with income between $30,000 and $34,999
Households with income between $35,000 and $39,999
Households with income between $40,000 and $49,999
Households with income between $50,000 and $59,999
Households with income >$60,000

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